2A97-34

STUDIES ON PACIFIC HERRING *Clupea pallasi* SPAWNING IN PRINCE WILLIAM SOUND FOLLOWING THE 1989 *EXXON VALDEZ* OIL SPILL, 1989-1992.

Draft Final Report to the *Exxon Valdez* Oil Spill Trustee Council for
Natural Resource Damage Assessment Fish/Shellfish Study Number 11

By

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and

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REGIONAL INFORMATION REPORT[1] NO. 2A97-34

Alaska Department of Fish and Game
Commercial Fisheries Management and Development Division, Central Region
333 Raspberry Road
Anchorage, Alaska 99518

June 97

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ACKNOWLEDGEMENTS

National Marine Fisheries Service (NMFS) personnel Jeffery Short, Stanley Rice, Adam Moles, Mark Carls, Sid Korn, and Chris Broderson (recognized experts in oil spill effects) provided advice, assistance, and guided sample collections for hydrocarbons and damage assessment research. NMFS personnel John Hamilton and Tom Rutecki participated in data collection in 1989. University of Alaska professors Michael Stekoll, Louis Halderson, Jeremy Collie, Theodore Cooney, and Brenda Norcross contributed advice and assistance in sampling design, data collection, and data analysis. Canadian Department of Fisheries and Oceans biologists Max Stocker, Jake Schweigert, Carl Haegele, Doug Hay and Ron Tanasichuk provided assistance in sampling design and data analysis. Alaska Department of Fish and Game personnel James Brady, Drew Crawford, Wayne Donaldson, Fritz Funk, Hal Geiger, Doug Jones, Steve Morstad, Sam Sharr, Keith Schultz and John Wilcock participated in project planning, implementation, and data collection. John Wilcock, Stephen Fried and Linda Brannian provided comments and suggestions during review of this report.

PROJECT SPONSORSHIP

This investigation was funded mainly by pre and post-settlement funds directly resulting from the Exxon Valdez oil spill fines as mandated by Federal Law included in the Natural Resource Damage Assessment Regulation of CERCLA.

Foreword

This document was originally produced in 1993 as a close-out summary of the Natural Resources Damage Assessment (NRDA) Fish and Shellfish Project Number 11 (F/S-11) funded by the *Exxon Valdez* oil spill Trustee Council (EVOS TC) from 1989 through 1992. Close-out reporting was necessitated by the 1992 Trustee Council decision to discontinue herring damage assessment studies. Unfortunately, this version of the close-out report was not the final version accepted by the Chief Scientist to complete the contractual obligations for the project, but it does contain unique and valuable information gathered during EVOS herring research that is not available in any other documents.

The F/S-11 project close-out report was prepared under considerable debate between program managers and project investigators concerning what form final reports for oil spill studies should take and who should pay for publishing costs. A number of documents were produced over the course of the herring damage assessment project prior to the 1992 close-out. Reports from individual subcontractors presented results for histopathology, cytology, cytogenetics, field observations of egg and larval mortality, and various experiments on exposures to oil. These contractor reports were produced as either final reports for discrete works or as annual progress reports for studies lasting more than one year. Titles and descriptions of these contractor reports were included in the F/S-11 project annual progress reports to the Trustee Council and, in some cases, contractor reports were appended to the project annual report. However, despite their importance in oil spill documentation, there was no clear direction for the handling of contractor work products in the EVOS reporting process and their presentation was inconsistent across years of EVOS herring studies. It was suggested early in the process of producing the 1989-1992 close-out report that all contractor reports be appended to the final version. This idea was rejected in favor of a single summary report that synthesized results from all project subcomponents.

The document presented herein was to be that synthesis of the project annual reports for 1989-1992, all contractor reports, and project results not previously reported. It was presented to the EVOS TC Chief Scientist in the summer of 1993, but was rejected on several grounds. In part, rejection was due to its length (118 pages). Reviewers felt that fecundity, egg loss, run timing, and diver calibration topics pertained to the spawn deposition methodology, but were not directly relevant to oil spill damage assessment and should be excluded from the close-out report. There were also substantive reviewer questions raised about the technical merits and validity of several components of this complex study including some of those pertaining directly to oiling. As well, the debate continued over how this report would relate to the subsequent publication of peer-reviewed journal articles by project investigators, when they would be produced, and who should pay for publication costs.

In the end, principal investigators and program managers agreed to a compromise consisting of eight draft journal articles prepared for publication in a special issue of the Canadian Journal of Fisheries and Aquatic Sciences (CJFAS) and an introductory chapter that would be a distillation of the original 118-page rejected draft. The subsequent version, submitted and accepted as the official F/S-11 project close-out final report, contained eight chapters (Table 1). A ninth chapter detailing histopathology results was never completed when publication dragged too long and the authors became embroiled in other topics. Chapter one, the introduction and drastically reduced

summary, was based on an early version of yet another notable document prepared for this project, an article for the proceedings of the 1993 EVOS symposium which was eventually published in 1996 (Table 2). The proceedings article summarized damage assessment work on early life history stages of herring, took a great deal of effort to finalize, and incorporated extensive oil related study results not available at the time of the symposium or the close-out decision. Production of the proceedings manuscript coincided with preparation of the F/S-11 close-out report and the journal articles, and added significantly to the general confusion regarding herring publications. Six peer-reviewed articles based on chapters from this version of the close-out report were eventually published in the October 1996 issue of the CJFAS (Table 3). Following symposium proceedings and journal article publication, the Chief Scientist officially recognized the completion of all project reporting obligations in a letter to Ms. Brown.

In response to reviewer criticisms, the accepted close-out document was focused more tightly on the effects of oil on herring than was the original report. However, reduction of the size of the original close-out report resulted in exclusion of a number of important analyses from any of the literature published by this project. Notable exclusions included two sections written by Tim Baker detailing an analyses of fecundity by area for all years and an analysis of egg loss data collected in 1990 and 1991. The egg loss section also contains information on distribution of eggs by depth for all years of the study. These sections were originally submitted to the project leader as draft status reports, but were incorporated directly into the single document by Evelyn Brown. Ms. Brown also wrote a section that was later excluded covering an analysis of mean spawn timing and run timing using historic data. A section on diver calibration modeling by several authors, but primarily based on Tim Baker's work, was also excluded.

The excluded analyses never received extensive review and some would benefit greatly from revision or the use of alternative analytical techniques. However, results of several of these analyses have received recurrent use by subsequent spawn deposition project leaders, as well as by other herring researchers within and outside the Department in spite of their basis on unpublished data. This absence from the citable literature has only allowed authors referring to these works to cite them as unpublished data, personal communications, or exclude the reference from their present work. In addition, rejection of the first version, subsequent acceptance of a revised version, and later publication of peer reviewed journal articles, in conjunction with the abundance of other documents produced through this research, has generated considerable confusion about what information is actually available in the official publications. The unpublished work in this document is being incorporated into the Commercial Fisheries Management and Development Division's Regional Information Report series at this time to assist researchers who wish to simplify use of the previously unpublished information in their present work, as a starting point for improvements to several of the data analyses, and to attempt to clear some of the confusion surrounding project F/S-11 documentation (albeit with the distribution of yet one more document).

Table 1. Chapters titles of the accepted 1993 final report to the EVOS Trustee Council for NRDA Fish/Shellfish project 11. Published version did not include chapter 9.

Chapter	Title	Authors
1	The *Exxon Valdez* oil spill and Pacific herring in Prince William Sound, Alaska: a summary of injury from 1989-1993 and recommendations for future inquiries.	E.D. Brown, T.T. Baker, F. Funk, J.E. Hose, R.M. Kocan, G.D. Marty, M.D. McGurk, B.L. Norcross, and J.W. Short
2	The distribution of oil spilled from the *Exxon Valdez*, ocean conditions affecting it, the initiation of damage assessment studies in response, and exposure of Pacific herring (*Clupea pallasi*) to oil in Prince William Sound.	E.D. Brown, B.L. Norcross, T.T. Baker, and J.W. Short
3	Effects of the *Exxon Valdez* oil spill on the survival of Pacific herring (*Clupea pallasi*) eggs.	T.T. Baker, M.D. McGurk, and E.D. Brown
4	Egg-larva mortality of Pacific herring in Prince William Sound, Alaska, after the *Exxon Valdez* oil spill.	M.D. McGurk and E.D. Brown
5	Sublethal effects of the *Exxon Valdez* oil spill on herring embryos and larvae: morphologic, cytogenetic, and histopathological assessments, 1989-1991.	J.E. Hose, M.D. McGurk, G.D. Marty, D.E. Hinton, E.D. Brown, and T.T. Baker.
6	Pacific herring (*Clupea pallasi*) embryo sensitivity to Prudhoe Bay petroleum hydrocarbons: laboratory evaluation and *in situ* exposure of embryos at oiled and unoiled sites in Prince William Sound.	R.M. Kocan, J.E. Hose, E.D. Brown, and T.T. Baker.
7	Larval herring distribution, abundance, and sublethal assessment in Prince William Sound, Alaska, following the *Exxon Valdez* oil spill.	B.L. Norcross, M. Frandsen, J.E. Hose, and E.D. Brown.
8	Reproductive success and histopathologic changes of individual Prince William Sound herring three years after the *Exxon Valdez* oil spill.	R.M. Kocan, G.D. Marty, E.D. Brown, and T.T. Baker.
(9)	Histopathologic analysis of chronic effects of the *Exxon Valdez* oil spill on Pacific herring in Prince William Sound, 1989-1991.	G.D. Marty, J.E. Hose, A.D. Moles, D.E. Hinton, and D.E. Hinton.

Table 2. Title of article summarizing EVOS herring oil spill research published in 1996 in the proceedings of the 1993 *Exxon Valdez* oil spill symposium.

Title	Authors
Injury to the early life history stages of Pacific herring in Prince William Sound, after the *Exxon Valdez* oil spill.	E.D. Brown, T.T. Baker, J.E. Hose, R.M. Kocan, G.D. Marty, M.D. McGurk, B.L. Norcross, and J.W. Short

Table 3. Titles of articles based on EVOS herring oil spill research close-out report chapters published in the Canadian Journal of Fisheries and Aquatic Sciences, Volume 53, October 1996.

Title	Authors
An introduction to studies on the effects of the *Exxon Valdez* oil spill on early life history stages of Pacific herring, *Clupea pallasi,* in Prince William Sound, Alaska.	E.D. Brown, B.L. Norcross, and J.W. Short
Egg-larval mortality of Pacific herring in Prince William Sound, Alaska, after the *Exxon Valdez* oil spill.	M.D. McGurk and E.D. Brown
Sublethal effects of the *Exxon Valdez* oil spill on herring embryos and larvae: morphological, cytogenetic, and histopathological assessments, 1989-1991.	J.E. Hose, M.D. McGurk, G.D. Marty, D.E. Hinton, E.D. Brown, and T.T. Baker.
Pacific herring (*Clupea pallasi*) embryo sensitivity to Prudhoe Bay petroleum hydrocarbons: laboratory evaluation and in situ exposure at oiled and unoiled sites in Prince William Sound.	R.M. Kocan, J.E. Hose, E.D. Brown, and T.T. Baker.
Distribution, abundance, and morphological condition, and cytogenetic abnormalities of larval herring in Prince William Sound, Alaska, following the *Exxon Valdez* oil spill.	B.L. Norcross, J.E. Hose, M. Frandsen, and E.D. Brown.
Reproductive success and histopathology of individual Prince William Sound herring 3 years after the *Exxon Valdez* oil spill.	R.M. Kocan, G.D. Marty, M.S. Okihiro, E.D. Brown, and T.T. Baker.

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EXECUTIVE SUMMARY

Pacific herring *Clupea pallasi* began migrating to spawning areas in Prince William Sound (PWS) only a few days after the *Exxon Valdez* oil spill. Commercial herring fisheries were closed in 1989 to avoid product contamination. Herring eggs and larvae are very sensitive to oil contamination, and a large percentage both herring spawning (40%) and rearing (over 80%) areas were within the trajectory of spilled oil. There was also concern about the short- and long-term effects of oil exposure on adult and juvenile herring. The Alaska Department of Fish and Game (ADF&G) became lead agency for herring studies since their staff had primary management authority for this resource and already had a suite of studies which could be modified to investigate oiling effects. The National Marine Fisheries Service (NMFS), which had staff with extensive experience examining effects of oil on marine organisms, assisted ADF&G staff in designing and implementing herring damage assessment studies. The 1989 damage assessment investigation consisted of expanded spawn deposition surveys, intensified age-weight-length and fecundity sampling, field and lab studies on eggs and larvae, histopathological sampling of adult herring, and hydrocarbon sampling of herring and mussel tissues. The overall goal of these studies was to describe injuries to PWS herring, including population effects, caused by the *Exxon valdez* oil spill. Only studies conducted by ADF&G staff are contained in the present report. Specific objectives of the ADF&G component during 1989-1992 were to: 1) estimate the extent of oil exposure by spawning herring adults and eggs from spill trajectory maps; 2) estimate spawning herring abundance from spawn deposition surveys each year of the study; 3) determine egg weight, gonad weight, and gonosomatic index (GSI) for female herring spawning in PWS each year of the study; 4) estimate PWS herring fecundity each year of the study; 5) estimate egg loss from spawn deposition sites during two years; and 6) quantify oil exposure at study sites from results of hydrocarbon analyses of mussel *Mytilus sp.* samples. Results of herring studies conducted by private contractors (e.g. larval herring surveys, egg and larval survival experiments, histopathology and cytogenetic investigations) are documented in the appendices of the summary report and other contractor reports (Biggs et al. *in prep*; McGurk 1990a, 1990b, 1991a, 1991b, 1991c, 1991d, and 1991e).

While only low dissolved oil concentrations were found in the water column, high concentrations of oil were found in mussel tissue at oiled study sites following the oil spill. Laboratory analyses showed that oil found in mussels was from the *T/V Exxon Valdez*. This suggested that the main route of oil exposure for herring was probably through contact with oil-water droplets or oiled particulate matter suspended in the water column. Nevertheless, dissolved oil concentrations in the water column may have been greater than those measured by the sampling techniques used. Actual concentrations may have approached the lower limit (10 ppb total aromatics) known to cause effects in marine organisms. Toxic micro-layers of surface oil, 10 to 10,000 times greater than oil concentrations several inches below the water surface, probably washed over herring eggs attached to intertidal areas. In 1989, total aromatic concentrations found in mussel samples in oiled areas ranged from 115.2 to 2,532.8 ppb. In 1990, oil concentrations in mussel tissue remained elevated in oiled areas, but levels of total aromatics decreased to 29.5 to 623.3 ppb. It was unlikely that previous contamination from other sources affected results of oil spill injury studies, since oil contamination in mussels from Rocky Bay, an oiled study site, prior to 1989 (1977 to 1980) was not from crude oil and was present at concentrations 10 to 1,000 times lower than found in mussels collected from this site in 1989.

Although levels of aromatic hydrocarbons were higher in both gut and gonad tissue of herring collected from oiled areas than from unoiled areas, amounts found were low. More convincing evidence of oil exposure in adult herring was shown by histopathological and parasite data. Severe hepatic necrosis,

consistent with exposure to a toxin, was found in 20% of herring from oiled areas but in no herring from unoiled areas. Furthermore, herring from oiled areas had much lower numbers of gut parasites than did herring from unoiled areas. To duplicate the effects observed in 1989 adult herring, herring held in a NMFS laboratory had to be exposed to total aromatic concentrations ranging from 0.68 and 1.20 ppm for 12 days. Therefore, while it is not possible to estimate the level or extent of exposure, it is probable that portions of the PWS herring population were exposed to patches of high concentrations of aromatic hydrocarbons in 1989.

Evidence of continued exposure to oil beyond 1989 was obtained from the bile of adult herring collected in the spring of 1990. Herring caught in both oiled and unoiled areas had whole fraction crude oil in their bile, indicating that herring spawning in unoiled locations had been exposed to oil sometime during their migration. However, herring sampled from unoiled areas in the fall of 1990 had much fewer histopathological signs of oil exposure than herring sampled in oiled areas. In addition, mixed-function-oxidase tests from these same fall-caught herring showed a greater induction of P450 (an indicator for oil exposure) in samples from oiled areas.

Although other studies have indicated that oil exposure could result in premature herring spawning, this type of effect was not obvious in Prince William Sound in 1989. While spawning activity in 1989 was the earliest recorded since 1973, this was probably influenced more by changes in ocean temperatures. Mean date of spawn is significantly correlated to winter and spring Gulf of Alaska temperature anomalies (ANOVA, $p < 0.05$). Although there were changes in spawn distribution and timing from 1989 to 1992, this variation could not be explained by the presence of oil. Similarly, while egg density within spawning areas was lowest in 1989 and greatest in 1991, oil effects were not obvious.

During the period 1988-1992, based on age structured analysis (ASA), the PWS herring population should have been fairly large in 1989 (95,404 tonnes) with a decrease in abundance over the next two years (66,093 tonnes in 1991) and an increase to a record high level in 1992 (121,684 tonnes). Actual estimates of herring abundance for this same time period based on egg deposition surveys, however, indicated that the population was much smaller in 1989 (45,281 tonnes), although overall trends were similar to that expected from ASA results. Other studies have shown that failure to spawn, due to reduced plasma levels of sex steroids, occurred after exposure to oil. Because we did not measure female hormonal levels, we could not determine whether the difference between the expected population size and the estimated actual size in 1989 was due to the oil spill. Egg retention and oocyte-loss measurements conducted in 1989 did not explain differences between egg deposition and ASA estimates. Temporal sampling of female gonads, which may have provided more clues, was not done.

Although differences in reproductive rates were found between oiled and unoiled areas, environmental factors may have affected fecundity, egg weight, and gonad weight of herring more than oil contamination. Fecundity, egg weight, and gonad weight were all linearly related to female body weight, but intercepts and slopes were significantly different among years and areas. Fecundity, egg weight, gonad weight, female body weight, and gonosomatic index all appeared to be strongly influenced by sea surface temperature anomalies several months prior to the time of spawning.

Since loss of herring eggs from spawning beds can greatly affect herring biomass estimates based on spawn deposition surveys, we conducted egg loss studies in 1990 and 1991 to increase biomass estimate accuracy and help us to detect oiling effects. Egg loss was greatest in areas with large amounts of intertidal spawning and reached a maximum at an intertidal depth of 1.65 m. Based on data from all areas and depths, overall daily egg loss was 5.4% in 1990 and 2.1% in 1991. However, since most spawning in PWS occurs in subtidal

areas, more realistic overall daily egg loss estimates of 4.0% in 1990 and 0.2% in 1991 were obtained by excluding data from the 1.65 m level. Since spawn deposition surveys were conducted 5 to 7 days after spawning had ceased, daily egg loss was estimated to be between 10.5% and 14.7% at the time of surveys. These egg loss estimates were similar to the 10% egg loss estimate already being used which had been based on studies conducted in Canada. Total egg loss during the 23.5 days of an average incubation period was estimated to be 50.4%, excluding 1.65 m depth estimates. Our two years of data indicated that eggs were eyed approximately 17.6 days from the spawning date.

Herring eggs, larvae, and adults were exposed to oil from the T/V *Exxon Valdez* tanker in 1989. Acute effects were measured only in the egg and larval stages and are detailed in the appendices of Biggs et al. (*in prep*). Sublethal effects were observed in all life stages and continued into 1990. Oil exposure and indices of larval sublethal damage were significantly correlated. The oil did not seem to affect adult herring spawning run timing and distribution. There may have be an oil effect on herring egg production, egg size, and spawning in 1989, but the results are masked by environmental influences or lack of specific data. Studies to measure long-term effects of oil exposure on the population of herring in PWS were not continued in 1993.

Recommendations for future studies include maintaining accurate stock assessment tools, continuing to document reproductive impairment in adults exposed to the oil in 1989 and 1990 through a laboratory study, resolving stock discreetness and distribution questions through a tagging study, conducting surveys on density, growth, and distribution of larval and juvenile herring, and implementing a hydroacoustic survey for following and sampling winter herring. Many of these projects could be coordinated with other forage fish studies and correlated with changes in abundance of the birds and mammals that rely on them for a food source. We recommend incorporating PWS herring studies and results into an ecosystem approach monitoring plan. We also recommend incorporating a summary of results and recommendations in a damage assessment portion of a response plan to streamline post-event study design in the case of the next oil spill.

KEY WORDS: herring, Pacific herring, *Clupea pallasi*, spawning, spawn deposition, Prince William Sound, hydrocarbons, *Exxon Valdez* oil spill

INTRODUCTION

Large numbers of Pacific herring *Clupea pallasi* annually migrate to spawn in nearshore areas of Prince William Sound (PWS) with the first schools usually observed in late March and early April (Figure 1). Herring are the target of large commercial fisheries as well as an important component of the marine ecosystem. In 1989, the spawning migration began only a few days after the *Exxon Valdez* oil spill. Various studies had shown oil contamination could have profound effects upon herring productivity, particularly through effects upon early life history stages. Commercial herring fisheries were closed in 1989 to avoid contamination of products with oil, and studies designed to determine the impact of oil on the herring population were rapidly implemented for Natural Resource Damage Assessment (NRDA) as mandated by the Comprehensive Environmental Response, Compensation, and Liability Act (CERCLA). Most existing studies documenting effects of oil on herring and other marine fish have come from controlled laboratory experiments using the water soluble fraction (WSF) of crude oil which contains the petroleum aromatic hydrocarbons (PAH). PAH are the most toxic components of crude oil, and some of these compounds are mutagenic (Connell and Miller 1984; Rice 1985). Unfortunately, results from laboratory experiments are often difficult to compare since concentrations and mixtures of PAH used have varied widely among studies (Rice 1985).

While herring gametes are vulnerable to oil pollution because they are incapable of detoxifying oil and because they are shed close to, and come in contact with, the water surface where the highest concentration of oil occurs (Rice 1985), fertilized eggs are also very sensitive to oil contamination. Sublethal concentrations of oil (below 3.0 ppm estimated total oil concentration) caused a majority of the surviving herring embryos observed to hatch prematurely or to be abnormal (Lindén 1978), while higher concentrations (e.g. 1.0 ppm PAH for 12 days) result in large reductions in hatching success with half the embryos dying at a PAH concentration of 1.5 ppm (Rice et al. 1987a). Herring embryos have been shown to accumulate eleven times the initial concentration of benzene in solutions (Kocan et al. 1987). Benzene has a narcotic effect in the embryos at low concentrations (0.06 ppm) and a stimulatory effect at high concentrations (0.56 ppm) which can lead to lowered growth (Eldridge et al. 1977). Yolk-sac larvae appear to be more resistent to changes in metabolism due to benzene exposure (Eldridge et al. 1977; Rice et al.). Herring larvae hatching from embryos exposed 12 hrs or more to crude oil solutions showed decreased growth that could have resulted from an impact on the mitochondria (Smith and Cameron 1979; Cameron and Smith 1980). Abnormal mitoses and other chromosomal abnormalities occur in fish eggs exposed to oil, particularly after treatment with weathered oil (Falk-Peterson and Lønning 1964). Most herring larvae hatching from embryos exposed to a WSF concentrations of 3 ppm either died or had abnormalities which would probably prove fatal (e.g. curved spines, abnormal or missing jaws, and swellings around the yolk sac and pericardium; Lindén 1978). Other investigators reported similar findings (Pearson et al. 1985).

Larval stages of marine fishes have been shown to be very sensitive to oil contamination in a number of laboratory studies (Mironov 1967,1969; Lindén 1974, 1975, 1976, 1978; Moore and Dwyer 1974; Struhsaker et.al. 1974; Kuhnhold 1977; Rice et. al. 1979; Smith and Cameron 1979; Rice et al. 1987a, 1987b). Fish larvae have been estimated to be 10 times more sensitive to PAH concentrations than adults (Moore and Dwyer 1974). However, oil effects observed in larvae are most profound if exposure occurs prior to rather than after hatching. Herring larvae have low survival rates even in unpolluted waters (Rice et al. 1987b) and are very vulnerable to surface oil (Rice et al. 1984). Some studies suggest that exposure for more than 16 hours could devastate an entire year class of herring (Rice et al. 1987b). Feeding larvae

have been killed during laboratory experiments by lower concentrations and shorter exposures to oil than either adults or embryos: growth was strongly inhibited by concentrations of WSF as low as 0.5 ppm, and feeding sharply declined, while mortality increased, in concentrations as low as 0.3 ppm (Rice et al. 1987b). However, while feeding in herring larvae is inhibited by dissolved aromatic hydrocarbons, larvae can depurate toxins rapidly and resume feeding if they are returned to clean seawater (Struhsaker et al. 1974).

Exposure of spawning adult herring to oil may also result in injury to embryos and larvae. In one study, the survival of spawned herring eggs was halved when prespawning females were exposed to very low levels of benzene (ppb) for 48 hrs (Stuhsaker 1977). Additionally, exposed females exhibited aberrant swimming behavior and spawned prematurely. In contrast, another study showed no increase in dead eggs or atretic follicles, and no decrease in egg size or hatching success when female spawning herring were exposed to 1.6 ppm PAH for 12 days, although some adult herring would die at this concentration of PAH (Rice et al. 1987b). However, adult herring can accumulate PAH rapidly in their ovaries. After 6 days of exposure to water containing 0.6 ppm PAH, adults contained 46.0 ppm PAH within their mature ovaries and were still accumulating PAH after 10 days of exposure (Rice et al. 1987a, 1987b). The investigators attributed this phenomenon to the presence of high lipid levels and an inability to metabolize oil when herring were in a reproductive state.

Significant changes in egg size and weight can also occur during stressful conditions such as an oil spill. Since egg is an indicator of available nutrients to the developing embryo, this has important implications for survival. However, natural environmental influences, such as water temperature, latitude, and food supply, also affect reproductive potential (Nikolsky et al. 1973), and must be considered when assessing oil spill effects. For example, an inverse relationship between latitude and fecundity has been described for selected North American herring populations (Paulson and Smith 1977), and both fecundity and egg size are significantly correlated with interannual variations in winter temperatures (Tanasichuk and Ware 1987). There may be an optimal egg size that maximizes survival, since egg weight was found to be inversely related to fecundity in White Sea herring populations (Lapin 1987).

Few papers are available which document effects of actual oil spills, particularly long term damages at the population level, upon marine organisms (Rice 1985; Karinen 1988). Most available information comes from areas with long-term chronic pollution problems, like San Francisco Bay, not from previously pristine areas like PWS. Documentation of oil spill effects on herring has mainly involved Atlantic herring in the North and Baltic Seas. Hatching success for Baltic Sea herring eggs (54%) during the four years prior to the *Tsesis* spill of 1977 was more than twice that found in oiled areas (25%) during the year of the spill (Aneer and Nellbring 1982). Sublethal effects on fish larvae following a spill in the Baltic were reported by Urho (1990). Histopathological evidence of oil exposure has been obtained from organisms as much as two years following the *Amoco Cadiz* oil spill (Haensly et al. 1982). Long term effects upon organisms have been inferred from results of laboratory studies and field collection, but have not been well documented. Fishes exposed to PAH produce mutagenic metabolites that bind to and damage both cellular DNA (including gonadal DNA) and proteins (Tan and Melius 1986; Varanasi 1989). Similarly, histopathological damage to internal organs and physiological effects in fishes exposed to oil have been well documented (Rice 1985; Karinen 1988). Many investigators (Rice et al.1984; Rice 1985; Karinen 1988) believe that such damages from oil exposure ultimately decrease growth, increase susceptibility to disease, and decrease reproductive success. All these effects will lower survival and long term production. However, decreases in fish stocks have never been attributed to oil spills, and extensive fish kills after oil spills have generally not been observed (Rice 1985).

The *Exxon Valdez* oil spill provided an excellent opportunity to document effects of oil on Pacific herring in the field and to compare them to effects observed during laboratory studies. Although larval survival is commonly cited as a critical determinant of adult year class strength (Lasker 1985; Smith 1985), we recognized that oil induced larval mortality would be hard to detect in the resulting adult population. Rice (1985) stated that an oil spill would have to kill over 50% of the larvae in a large population to detect this when the affected year class returned as adults. Therefore, we felt it would be more fruitful to turn our efforts to studying the effect of the oil spill on spawning herring and their progeny, since spawning herring are sensitive to petroleum compounds and oiling effects are also manifested in resulting eggs (Struhsaker 1977).

Several agencies and individual experts participated in designing studies to measure injury to PWS herring due to the *Exxon Valdez* spill. The Alaska Department of Fish and Game (ADF&G) was chosen as the lead agency for herring damage assessment studies because their staff had investigated herring in PWS for over 20 years and had baseline data on which to build population studies. National Marine Fisheries Service (NMFS), Auke Bay Laboratory staff, who had 10 years of experience researching effects of oil on marine organisms, assisted ADF&G in designing herring studies and provided advice on collecting samples for oil indexing. Herring experts from the University of Alaska (UA) and the Canadian Department of Fisheries and Oceans (DFO) were also consulted on study design.

The overall goal of the herring study was to describe any injuries to PWS herring, including population effects, caused by the *Exxon valdez* oil spill. Specific objectives of the component conducted by ADF&G staff during 1989-1992 and covered by this report were to: 1) estimate the extent of oil exposure by spawning herring adults and eggs from spill trajectory maps; 2) estimate spawning herring abundance from spawn deposition surveys each year of the study; 3) determine egg weight, gonad weight, and gonosomatic index (GSI) for female herring spawning in PWS each year of the study; 4) estimate PWS herring fecundity each year of the study; 5) estimate egg loss from spawn deposition sites during two years; and 6) quantify oil exposure at study sites from results of hydrocarbon analyses of mussel *Mytilus sp.* samples. Results of herring studies conducted by private contractors (e.g. larval herring surveys, egg and larval survival experiments, histopathology and cytogenetic investigations) were documented in previously submitted reports (Appedices in Biggs et al. in prep;McGurk 1990a, 1990b, 1991a, 1991b, 1991c, 1991d, and 1991e).

OBJECTIVES

Herring study objectives for each year were listed in detailed plans provided to the Trustee Council through the Natural Resource Damage Assessment (NRDA) process. While the overall goal of the study never changed, specific objectives were sometimes modified as information gaps were identified and when previous results indicated the need for more information in a particular area.

1989 NRDA Study Plan

In 1989, the year of the *Exxon Valdez* oil spill, most investigators thought that the "oil spill would adversely impact adult fish through direct mortality, food shortages, slowed growth, and a possible reduction in fecundity" while sublethal effects could include "reduced egg survival, reduced hatching success, and reduced viability of fry" (Trustee Council 1989). The following six objectives were included in the 1989 study plan to explore some of these effects (Trustee Council 1989):

1) Expand the normal sampling of herring populations in PWS to increase the precision of herring abundance, age composition, weight, sex ratio, and fecundity estimates. Specifically we intend to:

   Estimate the biomass of the spawning stock of herring in PWS such that the estimate is within ± 25% of the true value 95% of the time;

   Estimate the age, weight, length (AWL), and sex composition of herring in PWS during 1989 such that age composition estimates are with ± 5% of their true values 95% of the time.

2) Document the occurrence of herring spawn in oiled and non-oiled areas.;

3) Estimate hydrocarbon contamination of, and physiological impacts on, adult herring in oiled and non-oiled areas. Specifically we intend to:

   Test the hypothesis that the level of hydrocarbons in herring tissues is not related to the level of oil contamination on the area from which the herring were sampled. The experiment is designed to detect a difference of 1.6 standard deviation in hydrocarbon content with the probability of making a type I and type II error of 0.05 and 0.1, respectively;

   Estimate the presence and type of damage to tissue and vital organs of herring sampled from oil-impacted and unimpacted areas;

   Test the hypothesis that the level of hydrocarbons in herring eggs is not related to the level of oil contamination of the area from which the herring were sampled. The experiment is designed to detect a difference of 1.6 standard deviations in hydrocarbon content with the probability of making a type I and type II error of 0.05 and 0.1 respectively.

4) Estimate the proportion of dead herring eggs in oiled and non-oiled areas;

5) Estimate the hatching success, viable hatch, and occurrence of abnormal larvae by collecting herring eggs from oiled and non-oiled areas and rearing them under laboratory observation.

6) Identify potential alternative methods and strategies for restoration of lost use, populations, or habitat where injury is identified.

Since UA and NMFS staff were already conducting *Exxon Valdez* oil spill oceanographic studies, we assumed that information about plankton availability, food shortages, and environmental conditions would be available from those organizations. Studies on injuries sustained by larval and juvenile herring were not included in the 1989 study, which in hindsight, we feel was a mistake.

1990 NRDA Study Plan

Planning for the 1990 field season was difficult. Measurable adult mortalities had not been documented, and the herring food supply appeared to be good since the 1989 zooplankton bloom was comparatively large. Although analysis of 1989 study data had not been completed, preliminary results influenced several of the objectives for the 1990 study plan. Objectives 1 and 2 from the 1989 study plan, which had been accomplished, were included within the 1990 plan. A new project component measuring herring egg loss from the date of spawn deposition to the date of the survey was added in 1990 as part of Objective 4 to reduce the error in estimates of total eggs deposited. Objective 3 from the 1989 plan was not met. Only one sample of adult herring tissue had been collected during the spring of 1989. However, collection of additional hydrocarbon and tissue samples was planned for 1990 in case some oil contamination occurred between the spring of 1989 and 1990 collection dates. Objective 4 was accomplished, and indicated 5% to 10% lower survival of eggs in oiled areas. This objective was continued in 1990 to document any changes in 1990. Objective 5 was accomplished and demonstrated reduced hatching success and larval viability, and increased levels of abnormalities and chromosomal breakage in larvae from oiled areas. Continue monitoring of sublethal effects was planned for 1990 determine whether they would change over time. Objective 6 for 1989 could not be reached since analyses were still preliminary and injuries could not yet be estimated. Therefore, no restoration options were recommended and this objective, felt to be premature, was removed from the 1990 study plan.

Since spawning areas were expected to have reduced petroleum hydrocarbon levels in 1990, effects of oil on eggs and larvae were expected to be less than those found in 1989. However, we did expect to see tissue injuries from crude oil exposure in 1989 in adults returning in 1990. Based on the 1989 preliminary results, the following six objectives were included in the 1990 study plan (Trustee Council, 1990):

1) Expand the normal sampling of herring populations in PWS to increase the precision of herring abundance, age composition, weight, sex ratio, and fecundity estimates. Specifically we intend to:

   Continue to estimate the biomass of the spawning stock of herring in PWS such that the estimate is within ± 25% of the true value 95% of the time;

   Estimate the age, weight, length (AWL), and sex composition of herring in PWS during 1990 such that age composition estimates are within ± 5% of their true values 95% of the time.

2) Continue to document the occurrence of herring spawn in oiled and unoiled areas, validating the sites with quantified oil level information obtained from shoreline survey maps and hydrocarbon analysis of 1989 and 1990 herring eggs and mussel tissue.

3) Continue to estimate the hydrocarbon contamination of, and physiological impacts on, adult herring by analyzing tissue samples:

   Test the hypothesis that the level of hydrocarbons in herring tissues is not related to the level of oil contamination on the area from which the herring were sampled. The experiment is designed to detect a difference of 1.6 standard deviation in hydrocarbon content with the probability of making a type I and type II error of 0.05 and 0.1, respectively;

   Estimate the presence and type of damage to tissue and vital organs of herring sampled from oil-impacted and un-impacted areas;

Test the hypothesis that the level of hydrocarbons in herring eggs is not related to the level of oil contamination of the area from which the herring were sampled. The experiment is designed to detect a difference of 1.6 standard deviations in hydrocarbon content with the probability of making a type I and type II error of 0.05 and 0.1 respectively.

4) Continue to estimate the proportion of dead herring eggs from a subsample of study sites in oiled and unoiled areas that were utilized in the 1989 egg mortality study, expanding the data base and providing sample sites for sample collection of live and preserved eggs. In addition, add an egg loss study at the egg mortality sites to increase the accuracy of the spawn deposition biomass estimates.

5) Continue to estimate the hatching success, viable hatch, and occurrence of abnormal larvae, and collect embryonic and larval tissue for sublethal testing including MFO, cytogenetics, RNA/DNA ratio analysis, and others by collecting herring eggs from egg mortality sites and rearing them under laboratory observation.

Since project reviewers felt that juvenile herring studies were not likely to provide demonstrable population level injuries due to oiling, no studies were implemented to look at effects of oil on juvenile herring. No larval fish work or plankton surveys were scheduled. This lack of information about early life history stages began to affect the completeness of damage assessment on herring in 1990.

1991 NRDA Study Plan

Although the 1991 planning process was hindered by incomplete results and ongoing data analysis, study objectives were based upon available results and trends from past years. Objective 1 of the 1990 study plan was met; the accuracy and precision of the spawn depostion estimate was kept within goals. Objective 2 was partially met by documenting the extent and location of herring spawn. However, no hydrocarbon results were available for collected mussel tissues to determine the level of oiling at each site and area. Both objectives 1 and 2 were continued in 1991 to add a third year of accurate baseline data. Hydrocarbon analyses of adult tissues collected in 1989 and 1990, necessary to fulfill Objective 3 from the 1990 study plan, were still incomplete. The lack of such oil indexing information continued to impair analysis and interpretation of damage assessment results, which in turn affected planning of the 1991 study. However, Objectives 2 and 3 included provisions to collect more hydrocarbon samples in 1991, in case this information proved to be important in final analyses. Lack of systematically sampled adult tissues in 1989 also did not allow a detailed histopathological analysis to meet Objective 3. However, preliminary results of a herring parasite study conducted by NMFS and a histopathological examination of 1989 adult herring both indicated that herring were exposed to a toxic substance prior to spawning in early April. Herring were most likely contaminated while traversing oil during migration. Tissues collected in 1990 continued to show histopathological differences between fish in oiled and unoiled areas indicating greater exposure of herring collected in oiled areas. Therefore, the collection of herring tissue for histopathological examination was slated to continue in 1991 to meet new Objectives 3 and 4.

Measurable differences in egg survival between oiled and unoiled areas continued in 1990 (Objective 4, 1990 study plan). However, the egg mortality survey in 1990 did not differentiate the damage due to direct exposure of embryos to oil at the study sites from indirect exposure of embryos to residual oil in the spawning adults. Therefore, some egg mortality in 1990

could have been from indirect exposure of embryos to oil rather than from exposure at the study sites. Injury to reproductive ability of the 1990 spawning adults that were exposed in 1989 was not measured. The overall egg survival rate was much lower in 1990 than in 1989 and was possibly due to high egg densities at the spawning grounds and not to residual oil at the spawning sites or residual effects in spawning adults. Objective 5 was continued in 1991 based on the hypothesis that a return to pre-spill baseline survival rates would be observed and to add a third year of results due to the natural variability in the data set.

Analysis revisions confirmed the 1989 preliminary results on injury to embryos and newly hatched larvae. In addition, the hypothesis that injury would lessen in 1990 was confirmed (part of Objective 5 in the 1990 study). Only small differences in morphological abnormalities and genotoxic injury were measured between oiled and unoiled areas in 1990. Differences in hatching success and viable hatch between oiled and unoiled areas, and a control area outside PWS, disappeared in 1990. MFO and RNA/DNA ratio data were not found to be useful in this analysis. Induction of MFO in herring is questionable and RNA/DNA ratios were not found to be significantly different even when sampled from larvae of varying size. Larvae from control areas in PWS and in Southeast Alaska were heavier and had larger yolk volumes than larvae from oiled areas in PWS. Based on the continued injuries observed, Objective 6 was included in the 1991 study plan to measure any persistent effects or return to baseline levels of abnormal larvae incidence in the third season after the spill. Confidence in the 10% estimate of egg loss used in calculations was boosted by results from the egg loss study in Objective 4 of 1990 study plan. The egg loss study was continued in 1991 to obtain two years of data as outlined in Objective 7 in the 1991 study plan.

Based on a decline in effects of oil on eggs and larvae and qualitative histopathological data, objectives for the 1991 study plan were as follows (Trustee Council 1991):

1) Expand the normal sampling of herring populations in PWS to increase the precision of herring abundance, age composition, weight, sex ratio, and fecundity estimates. Specifically we intend to:

   Continue to estimate the biomass of the spawning stock of herring in PWS such that the estimate is within ± 25% of the true value 95% of the time;

   Estimate the age, weight, length, sex (AWLS) composition of herring in PWS during 1990 such that age composition estimates are within ± 5% of their true values 95% of the time.

2) Document the occurrence of herring spawn in oiled and unoiled areas, validating the sites with quantified oil level information obtained from shoreline survey maps and hydrocarbon analyses of 1989, 1990, 1991 herring eggs and mussel tissue.

3) Estimate the hydrocarbon contamination of, and physiological impacts on, adult herring by analyzing tissue samples:

   Test the hypothesis that the level of hydrocarbons in herring tissues is not related to the level of oil contamination on the area from which the herring were sampled. The experiment is designed to detect a difference of 1.6 standard deviation in hydrocarbon content with the probability of making a type I and type II error of 0.05 and 0.1, respectively;

4) Estimate the presence and type of damage to tissue and vital organs of herring sampled from oiled and unoiled areas;

Test the hypothesis that the level of hydrocarbons in herring eggs is not related to the level of oil contamination of the area from which the herring were sampled. The experiment is designed to detect a difference of 1.6 standard deviations in hydrocarbon content with the probability of making a type I and type II error of 0.05 and 0.1 respectively.

5) Estimate the proportion of dead herring eggs in oiled and unoiled areas from a subsample of study sites that were utilized in the 1989 and 1990 egg mortality studies, expanding the data base and providing sample sites for sample collection of live and preserved eggs.

   Test the hypothesis that the proportion of dead herring eggs is not related to the level of oil contamination of the area from which the herring were sampled.

6) Estimate the hatching success, viable hatch, and occurrence of abnormal larvae, and collect embryonic and larval tissue for sublethal testing including cytogenetics, MFO analysis, and histopathological analyses by collecting herring eggs and rearing them under laboratory observation.

   Test the hypothesis that hatching success, viable hatch, and occurrence of abnormal larvae are not related to the level of oil contamination of the area from which the herring were sampled.

7) Estimate the number (proportion) of eggs removed from the spawning areas (due to wave action or predation) between the time of egg deposition (spawning) and the time of hatching.

1992 NRDA Study Plan

Objective 1 of study plan was accomplished, providing accurate herring spawning biomass estimates for three consecutive years. Locations and extent of spawn had also been accurately documented. We proposed to continue this objective as a restoration monitoring project for 1992.

Objective 2 from the 1991 study plan could not be completed because detailed oil indexing data were still unavailable. The State Department of Natural Resources (DNR) had not finished compiling trajectory and shoreline oiling information and NMFS hydrocarbon data had not yet been interpreted. Completion of this objective was proposed in the 1992 study plan.

Objectives 3 and 4 contained components that were in various states of completion. Histopathological analysis of 1991 adult herring tissue was still incomplete by the end of 1991. A fecundity analysis (not originally included as a 1991 objective) had been conducted but not fully completed. Oocyte analysis of developing ovaries in 1990 and 1991 did not reveal atrophied eggs or other injuries. However, it was later that temporal stratification was not sufficient in 1990 and 1991, and was lacking in 1989, to detect female reproduction injuries during those years. Parasite loads in 1991 adults had returned to baseline levels, and no differences were observed between herring from oiled and unoiled areas. Although adult herring tissue samples obtained three weeks after the spill contained only in low levels of petroleum hydrocarbons, 1989 adult herring bile samples revealed oil levels that corroborated injuries observed in the 1989 tissues as did the lack of parasites in oiled herring. A synthesis of this data set was proposed in the 1992 study plan.

Objective 5 was met and suggested that an overall increase in egg mortality in 1991 was probably due to severe environmental conditions rather than oil.

Similarly, differences in estimated egg loss between 1991 and 1990, measured to satisfy objective 7, were also probably due to severe weather conditions rather than oil.

Objective 6 was satisfied, and indicated no differences in rates of sublethal effects for embryos and larvae between oiled and unoiled areas. While estimated survival rates from hatching to free-swimming larvae were depressed in oiled areas in 1989, differential egg distribution, environmental factors, and survey design made it difficult to detect oiling effects. A synthesis of this data set was proposed in the 1992 study plan. The dose-response laboratory study and field exposure experiment were completed and provided detailed results to compare with 1989 and 1990 field results. Unfortunately, collected of free-swimming larvae in 1991 did not follow specified study plan methods, so interpretation of the results and direct comparisons with sublethal rates found in free-swimming larvae collected in 1989 had to be done carefully. A summary of the lab data set was proposed in the 1992 NRDA study plan.

Objective 7 was met and provided egg loss estimates for two consecutive years. A third and final season of egg loss measurements were proposed as part of a restoration monitoring project for 1992.

The 1991 status report did contain recommendations for potential restoration work and for the 1992 damage assessment study plan, but this objective was inadvertently left out of the published 1992 plan.

Two proposals were submitted for 1992: one for the completion of NRDA work (the 1992 NRDA study plan) and a second covering potential restoration work (the 1992 restoration monitoring study plan). Information gaps in data collection and tasks were identified and addressed in the restoration proposal. Two basic categories of information were identified as being necessary for completion of a population simulation model that would define present and future injury to PWS herring. The first category defined herring stock distribution, discreetness, and migration rates; the second defined processes affecting larval growth, survival, distribution, and recruitment. Unfortunately, only the NRDA proposal was approved in 1992, so information gaps identified in the restoration proposal remain.

Objectives identified in the 1992 NRDA study plan were as follows (Trustee Council 1992):

1) Estimate the total level of damage of the EVOS to the early life stages by:

   a. summarizing and synthesizing components of the egg mortality, egg incubation, and egg and larval cytogenetic and histologic examinations;

   b. summarizing the larval herring distribution and abnormality index data from the 1989 larval trawl survey;

   c. finalizing chemistry data from the hydrocarbon sample database;

   d. combining components a., b., and c. to relate level of oiling with level of damage.

2) Summarize the results from the laboratory and field exposure dose-response studies and to compare effects of known dosing on egg survival, hatching success, percent viable hatch, and larval abnormalities (Graded Severity Index), cytogenetics, and mixed function oxidase (MFO) levels to field data collected in 1989-1991. This data will be used to refine Objective 1.

3) Complete the literature review and compare results from other studies to the findings in Objectives 1 and 2.

4) Estimate the total level of damage to herring at the adult stage by:

   a. summarizing and synthesizing the histopathological presence and type of damage to tissues and vital organs from herring collected in oiled and non-oiled areas during 1989, 1990, and 1991;

   b. summarizing the level of egg atrophy in adult female gonads (oocyte-loss) in samples collected during 1989, 1990, and 1991;

   c. coordinate with National Marine Fisheries Service (NMFS/NOAA) to synthesize the results from the adult dose-response experiment (1991 and 1992), the adult parasite study (comparing herring from oiled and unoiled area during 1989 and 1991), and from other studies reported in the scientific literature.

5) Incorporate the results from Objectives 1 through 4 in a modeling effort that will utilize age-structured population information, environmental data, and hydrocarbon data to explore the overall effects of EVOS on herring in PWS.

1992 NRDA Study Plan Status

Objectives 1 through 4 were completed, with the exception of the NMFS adult dose-response study. Results of studies conducted by ADF&G staff are presented within this report, while results of studies conducted by various subcontractors are contained in separate reports (Hose 1993; Kocan and Mehl 1993; Norcross and Frandsen 1993; Marty et al., 1993; and McGurk 1990a, 1990b, 1991a, 1991b, 1991c, 1991d, and 1991e). Objective 4, total level of damage, could only be qualitatively described since critical information gaps were left by the disapproval of the 1992 restoration proposal as well as inadequate sampling of adult tissue in 1989 and 1990. However, early in 1992, after project plans had been approved, a pilot reproductive impairment study was devised to address objective 4 quantitatively. This study was designed to measure long-term effects of oil on the ability of PWS herring to produce viable offspring. It was hoped that results of this study would partially or fully mitigate the lack of conclusive and quantitative adult injury information. Findings of this pilot study indicated potential reproductive impairment in adult herring in oiled areas. However, because this study was not continued in 1993, conclusive results on long-term effects of oil on reproduction of herring are not available. Additionally, a research agreement was finally approved, after a one year administrative delay, which allowed larval herring collected in 1989 by UA to be examined. Results of this study provided important knowledge about herring life history between the early life stages and adults (Norcross and Frandsen 1993).

Finally, Objective 5 was partially completed. Models were constructed which synthesized early life stage and oiling, year class strength and recruiting processes, and stock abundance information. However, several key pieces of information were still missing, and a model estimating long term injury to PWS herring could not be completed.

A spawn deposition survey, providing an estimate of the spawning biomass, was conducted in 1992 even though it was not included in 1992 study plan objectives. While project funds for this survey under restoration monitoring were denied only days prior to the scheduled 1992 field season start-up date, a research vessel and personnel had already been secured in advance of project

approval to satisfy administrative procedures. Fortunately, sufficient oil spill damage assessment and State funding was available to allow the survey to be conducted. Results provided a fourth year of data needed to map population trends, provide a basis for population modeling, and supply samples needed to fulfill 1992 study plan objectives.

METHODS

Description of Oil Trajectory

Information on the trajectory and concentration of *Exxon Valdez* crude oil came from a variety of sources. During much of the four year NRDA herring study, 1989 maps produced by National Oceanographic and Atmospheric Administration (NOAA), Alaska Department of Environmental Conservation (DEC), and ADF&G aerial surveyors were the only sources of oiling information available. In 1991, detailed shoreline oil indexing information became available from DNR (GEO, Damage Assessment Geoprocessing Group 1991; Mortenson et al. 1993). Although DNR information was qualitative and not useful analysis of injury, the large, colorful maps produced were descriptive and useful in research planning and interpretation. The trajectory produced by the DNR On-Scene Spill Model (OSSM), developed by NOAA, depicted the main path of the oil on the water surface, but it did not represent a realistic subsurface distribution, since it did not account for oil-water droplets formed by high energy transfer due to waves and storms (Neff 1990), and it did not incorporate all available hydrocarbon chemistry results.

For our herring study, we used a combination of maps to represent the oil path: overflight maps produced by NOAA in 1989, composite maps produced by DEC (Marshall Kendziorek, personal communication, December 26, 1989), and a composite map received from the NMFS Auke Bay Laboratory in the fall of 1992. While we tried to corroborate the oil path with available chemistry data, difficulty in reaching consensus on the extent and nature of the oil trajectory reflects the difficulty of identifying the distribution of a partially soluble chemical complex in a liquid medium. This is compounded in PWS by local currents, numerous storm events, and convoluted shorelines.

A visual description of the oil trajectory in PWS was needed to determine oiled and unoiled study areas as well as the overlap between oil and herring distribution. We used a simple method to calculate the percentage of spawning areas in oiled areas: spawning areas inside or adjacent to the composite oil path were classified as oiled, while spawning areas entirely outside of the oil path were identified as unoiled. Level of oiling was determined entirely by hydrocarbon chemistry results from the mussel data set (Short and Heintz 1993).

Spawn Deposition Survey and Biomass Estimation

Management of the PWS herring stock is based on a harvest policy established by the Alaska Board of Fisheries which specifies a maximum 20% exploitation rate for the combined harvest of all herring fisheries (5 AAC 27.365; ADF&G 1993). The allowable harvest is based on biomass estimates established the previous year modified by the expected growth and survival over the year. Aerial surveys were used to estimate biomass from 1973-87, while spawn deposition surveys, first done in 1983 (Jackson and Randall 1983) and 1984

(Jackson and Randall 1984), began to be used as the primary biomass estimate in 1988 (Biggs and Funk 1988). Aerial surveys are easier to perform than spawn deposition surveys, but aerial survey biomass estimates are not as reliable since herring school visibility varies, residence time of herring schools on the spawning grounds is unknown, and neither accuracy nor precision estimates are available. ADF&G has continued to conduct annual aerial surveys of spawning biomass to provide information on run and spawn timing as well as school and spawn distribution and extent.

Biomass estimates derived from spawn deposition surveys, termed escapement biomass, measure numbers of herring which escaped commercial spring herring fisheries. The total herring population returning in the spring to spawn in PWS is termed run biomass and is equal to the commercial catch plus the escapement biomass estimate. A forecast, estimating run biomass for the following year, is calculated annually from the escapement biomass.

The existing spawn deposition program was modified in 1989 for NRDA studies to more accurately assess the PWS herring stock's response to the oil spill. Prior to NRDA studies, herring spawn deposition survey were designed to obtain biomass estimates within ± 28% of the true biomass 95% of the time. This level of precision was acceptable for estimating exploitation rates and forecasting future abundance. If weather or other logistic problems hampered spawn deposition surveys, accuracy and precision would be reduced. Beginning in 1989, spawn survey design was altered to obtain biomass estimates within ± 25% of the true biomass 95% of the time. This included increasing the number of 1) skiff surveys used to delineate spawning area boundaries; 2) survey transects, and 3) SCUBA divers. Skiff surveys not only increased the level of precision but also provided documentation of herring spawn in oiled and unoiled areas. More SCUBA divers ensured that the required number of transects were made during the brief two-week sampling window.

Biomass estimation based on spawn deposition surveys consisted of three major components: 1) a spawn deposition survey; (2) age-weight-length (AWL), sex ratio, and fecundity sampling; and (3) egg loss determination.

Spawn Deposition Survey Design

Survey design has been described in detail by Biggs and Funk (1988), and follows closely the two-stage sampling design of similar surveys in British Columbia (Schwiegert et al. 1985) and Southeast Alaska (Blankenbeckler and Larson 1982, 1987). Our surveys used random sampling for the first stage (transects) and systematic sampling for the second stage (quadrats within transects). Random sampling for the second stage was not feasible because of underwater logistical constraints (Schwiegert et al. 1985). In addition, our surveys were stratified by area to account for geographic differences and the potential for discrete herring stocks (Southeast, Northeast, North Shore, Naked Island and Montague; Figure 1).

Mean egg densities along each transect were combined to estimate an average egg density by area. Spawning bed width along each of the transects was used to estimate average spawning bed width by area. Average width, average density, and total spawning bed shoreline length were used to estimate total number of eggs deposited in each of the five areas. Average fecundity and sex ratio, derived from AWL sampling, and estimates of total number of eggs deposited were used to calculate herring population numbers and biomass. Based on variances obtained during the 1988 survey, 160 transects were needed to insure that estimated biomass would be within 25% of the true biomass 95% of the time. Confidence intervals were calculated assuming a normal distribution of total egg estimates.

Spawn Deposition Survey Sampling Procedure

The general location of spawning activity was determined from milt observed during scheduled aerial surveys. This information was compiled and summarized on maps showing spawning locations and the number of days on which milt was observed.

Using aerial survey maps, skiff survey crews in 1989 and 1990 visited each spawning area and determined the boundaries of areas actually containing herring eggs. The skiff survey team began mapping several days after spawning ended to ensure that all egg containing areas were recorded. In 1989, skiff surveys were conducted from a chartered salmon gillnet vessel with shallow draft capabilities and a jet drive. In 1990, skiff surveys were conducted from dive skiffs by the dive team immediately preceding the dive survey. Skiff surveys were performed close to shore at low tide by both walking along exposed intertidal areas and using an underwater viewing box, grappling hooks, and rakes from a small inflatable boat. Boundaries of herring spawning areas were recorded on maps. Total linear miles of shoreline containing herring spawn was estimated from skiff survey maps during 1989 and 1990, and from aerial survey maps during 1991 and 1992.

Diving on herring spawn was deferred until about 5 days after spawning had ceased to allow both water turbidity due to milt to decrease and the large numbers of sea lions were usually present near spawning herring to disperse. Two three-person diving teams were used to complete surveys. Each team consisted of a lead diver to count eggs, typically the most experienced in this survey technique, a second diver to record data, and a third diver that remained on the surface in a tender. Diving and tending duties were rotated daily. Each team worked either a morning or afternoon shift each day. Based on information from the 1988 PWS survey and Southeast Alaska surveys, two diving teams could complete 10 to 20 transects each day in favorable weather conditions and in areas with average spawning density and distribution. (In 1989, the northwest shoreline on Montague Island received thick spawn spread over a large area, so only 6 transects could be completed each day.) A total of 160 or more transects were needed each season, which, depending on weather and extent of spawn, required 15 to 25 days of diving.

Each shoreline area containing herring spawn was divided into the smallest resolvable segments on the map scale (0.1 mi or less). A total of 160 shoreline segments were selected at random from among all shoreline segments containing spawn for 1989-1992. Each transect was assigned a sequence number and charted on waterproof field maps. Exact transect location was fixed by having the dive team leader chose a shoreline feature (tree, rock, cliff, etc.) located above the high tide line within the randomly selected shoreline segment to use a reference point. This was done as the dive skiff approached the shore, but before bottom profiles, bottom vegetation, or herring spawn were visible from the skiff.

The sampling quadrat consisted of a 0.1 m^2 frame constructed of PVC pipe with a depth gauge and compass fastened to it. Data were recorded using a large weighted carpenter's pencil and data forms printed on water-proof plastic paper attached to a PVC clipboard.

Sampling along each transect occurred in the following manner:

1. A compass course perpendicular to the shoreline at the transect location was set on a compass attached to the sampling quadrat.

2. The first quadrat was haphazardly placed within the first 5 meters of spawn by tossing the sampling quadrat.

3. The lead diver measured four complete 1 m hand-spans while moving offshore along the compass course. Halfway through the fifth hand-span, the lead diver placed the sampling quadrat ahead approximately one-half meter in direct line with the compass course and estimated the number of eggs at the new quadrat location.

4. The lead diver estimated the number of eggs in units of thousands (K) in the quadrat and communicated his estimate through hand signals to the second diver who recorded it. Vegetation type, percent cover, substrate, and depth were also recorded.

5. This process was repeated every 5 meters until the apparent end of the spawn was found. Divers verified the end of the spawn by swimming at least an additional 20 m past the end of the spawn or until a steep drop-off was encountered or vegetation ended.

Becker and Biggs (1992) documented methods used for diver surveys including sample data forms, key codes for vegetation types, standard operating procedures for ADF&G diving, chemical recipes for sample preservatives, and other practical information.

Diver calibration samples were collected at approximately every fifth quadrat placement. Both divers estimated the number of eggs in each calibration quadrat independently and then attempted to collect all egg-containing vegetation as well as all eggs from rock substrates within the quadrat. Vegetation and eggs were placed in numbered mesh bags. The number of eggs left after the removal process was completed was estimated by the lead diver and recorded. Based on 1988 survey results, 80 calibration samples were needed for each diver, including 20 samples for each of four vegetation categories: eelgrass (EEL), fucus (FUC), large brown kelp (LBK), and hair kelp (HRK). Calibration samples were also stratified over three ranges of egg densities: low (0-20,000), medium (20,000-80,000), and high (>80,000). The spawn deposition project leader tracked the number of samples collected by each diver by vegetation group and density to ensure that sufficient calibration samples were taken in each category. Upon completing a dive shift, calibration samples were removed from numbered mesh bags, placed in nalgene ziploc bags, and completely immersed in Gilson's solution (Becker and Biggs 1992). A mylar label containing the transect number, date, both diver's estimates, and vegetation type was placed in the bag with the sample. Calibration sample bags were stored in sealed and numbered 5 gal plastic buckets.

Plastic buckets were processed in the laboratory in the Cordova ADF&G office using the following procedures:

1) Gilson's solution was decanted from the sample bags.

2) KOH was added to each sample bag and thoroughly mixed with the sample. The sample was soaked for approximately 1.5 hours depending on vegetation type in a hot water bath to accelerate digestive hydrolysis. Eelgrass and fucus samples required longer than hair kelps and LBK. The digestive process was monitored by gently manipulating the sample bags to help judge the soaking time required.

3) The KOH was decanted, egg loss was estimated, and the sample was poured into a 1 gal plastic bucket.

4) The sample was poured into an appropriately sized sieve and repeatedly rinsed with cold water to loosen attached eggs and to wash the dissolved kelp through the mesh.

5) Any eggs left attached to kelp were removed from the substrate by careful manual scraping. Loose eggs were cleaned of debris for accurate volumetric analysis. Eggs lost during cleaning were estimated and recorded.

6) The loose, clean eggs were poured back in labeled and rinsed sample bags and enough 1.0 Normal bufferred formalin saline solution was added to cover the eggs generously. The eggs were allowed to soak in the saline for 24 h to ensure standardized volumetric displacement.

7) The standard displacement of 1,000 eggs was determined for each sample. A subsample of 1,000 eggs from the sample was hand counted and the volume was measured in a 10 ml graduated cylinder.

8) The volume of the whole sample was measured and the total number of eggs in the sample was estimated by simple proportion.

Recipes to prepare Gilson's solution, as well as for other chemicals used to process samples, are described in Becker and Biggs (1992).

Spawn Deposition Survey Data Analysis

Biomass Estimation. Analysis of 1989 spawn deposition survey data was similar to methods used in 1988 (Biggs and Funk 1988). The biomass estimator was:

$$B = TB'; \quad (1)$$

where:

B = estimated spawning biomass in tonnes,
T = estimated total number of eggs (billions) deposited in an area, and
B' = estimated tonnes of spawning biomass required to produce one billion eggs.

Estimates for T and B' were derived from separate sampling programs and were thus independent. The estimated variance for the product of the independent random variables T and B' was (Goodman 1960):

$$Var(B) = T^2 Var(B') + B^2 Var(T) - Var(T) Var(B'); \quad (2)$$

where,

Var(B') = an unbiased estimate of the variance of B', and
Var(T) = an unbiased estimate of the variance of T.

Total Number of Eggs (T). The total number of eggs deposited in an area was estimated from a two-stage sampling program with random sampling at the primary stage, followed by systematic sampling at the secondary stage, using a sampling design similar to that described by Schwiegert et al. (1985). To compute variances based on systematic second stage samples, it was assumed that eggs were randomly distributed in spawning beds with respect to the 0.1 m^2 sampling unit. While this assumption was not examined, in practice the variance component contributed by the second sampling stage was much smaller

than that contributed by the first stage, so violation of this assumption would have little effect on the overall variance. The total number of eggs (T), in billions, in an area was estimated as:

$$T=N\hat{y}10^{-6}/(1-R); \quad (3)$$

where:

L	=	the shoreline length of the spawn-containing stratum in meters,
N	=	$L/0.1^{0.5}$ = the total number of possible transects,
$0.1^{0.5}$	=	0.3162 m = width of transect strip,
$\hat{y}$	=	average estimated total number of eggs (thousands) per transect,
10^{-6}	=	conversion from thousands to billions of eggs, and
R	=	estimated proportion of eggs disappearing from the study area from the time of spawning to the time of the survey.

Average total number of eggs per transect strip (in thousands) was estimated as the mean of the total eggs (in thousands) for each transect strip using:

$$\hat{y}=\frac{\sum_{i=1}^{n}\hat{y}_i}{n}; \quad (4)$$

where:

$$\hat{y}_i=M_i\bar{y}_i;$$

and:

n	=	number of transects actually sampled,
i	=	transect number,
M_i	=	$w_i/0.1^{0.5}$ = number of possible quadrats in transect i,
w_i	=	spawn patch width in meters measured as the distance along the transect between the first quadrat containing eggs and the last quadrat containing eggs, and
$\bar{y}_i$	=	average quadrat egg count in transect i (in thousands of eggs).

Average quadrat egg count within a transect, $\bar{y}_i$, was computed as:

$$\bar{y}_i=\frac{\sum_{j=1}^{m_i}y_{ij}}{m_i}; \quad (5)$$

where:

j	=	quadrat number within transect i,
m_i	=	number of quadrats actually sampled in transect i, and

y_{ij} = adjusted diver-estimated egg count (in thousands of eggs) from the diver calibration model for quadrat j in transect i.

The variance of T, ignoring the unknown variability in R, is similar to that given by Cochran (1963) for three stage sampling with primary units of equal size. In this case the expression is modified because the primary units (transects) do not contain equal numbers of secondary units (quadrats), and the variance term for the third stage comes from the regression model used in the diver calibration samples. Therefore the estimated variance of T, conditioned on R, is:

$$Var(T) = \frac{[N^2(10^{-6})^2[\frac{(1-f_1)}{n}s_1^2 + \frac{f_1(1-f_2)}{\sum_{i=1}^{n} m_i}s_2^2 + \frac{f_1 f_2}{\sum_{i=1}^{n} m_i}s_3^2]]}{(1-R)^2}; \quad (6)$$

where:

$$s_1^2 = \frac{\sum_{i=1}^{n}(\hat{y}_i - \bar{\hat{y}})^2}{n-1} = \quad (7)$$

variance among transects,

$$s_2^2 = \sum_{i=1}^{n} M_i^2 \sum_{j=1}^{m_i} \frac{(y_{ij} - \bar{y}_i)^2}{n(m_i - 1)} = \quad (8)$$

variance among quadrats,

$$s_3^2 = \sum_{i=1}^{n} \sum_{j=1}^{m_i} Var(y_{ij}) = \quad (9)$$

sum of the variances of the individual predicted quadrat egg counts from the diver calibration model,

$$f_1 = \frac{n}{N} = \quad (10)$$

proportion of possible transects sampled, and

$$f_2 = \frac{m_i}{M_i} = \quad (11)$$

proportion of quadrats sampled within transects (same for all transects).

Diver Calibration. Diver observations of vegetation species were aggregated into four vegetation categories based on structural and phylogenetic similarities of plants in the quadrat: eelgrass, fucus, hair kelp, and large brown kelp (Becker and Biggs 1992). Diver estimates of egg numbers were proportional to laboratory-enumerated counts, but systematic biases in the diver estimates were accounted for by vegetation type and density (Biggs and Funk 1988). Although individual diver effects were not significant in the 1988 survey, potential differences among individual divers was examined for NRDA surveys. The basic form of models used to account for biases in diver observations was:

$$Y_{ijk}=e^{\alpha}e^{D_j}e^{V_k}X_{ijk}^{\beta_{jk}}e^{\epsilon}; \qquad (12)$$

where,

- α = a constant,
- D_j = parameters representing the effect of the j^{th} diver,
- V_k = parameters representing the effect of the k^{th} vegetation type,
- $ß_{jk}$ = parameters controlling the functional form of the relationship between the diver estimate and laboratory-enumerated egg count for diver j in vegetation type k,
- Y_{ijk} = the i^{th} laboratory egg count in the vegetation-diver stratum jk,
- X_{ijk} = the i^{th} diver estimate in vegetation-diver stratum jk, and
- ϵ = a normally distributed random variable with mean 0 and variance σ^2.

A multiplicative-effect model was chosen because relative estimation errors were expected to change with egg density. The distribution of laboratory-enumerated egg counts for a given diver estimate was positively skewed in the 1988 survey (Biggs and Funk 1988), so that the logarithmic transformation used to estimate the parameters of the multiplicative-effect model also stabilized the variance and corrected the skewness of the egg density estimates. After a logarithmic transformation equation 12 becomes:

$$\log_e(Y_{ijk}) = a + D_j + V_k + \beta_{jk}\log_e(X_{ijk}) + \epsilon; \qquad (13)$$

where,

- $ß_{jk}$= the slope of the relationship between the logarithm of the diver estimate and the logarithm of the laboratory-enumerated egg count.

In logarithmic form, the model comprises a linear analysis of covariance problem with two factor effects (vegetation and diver) and 1 covariate (diver-estimated egg number). The SAS (1987) procedure for general linear models was used to obtain least squares estimates of parameters and evaluate variance components. In addition to the two factor effects and one covariate, terms for diver-vegetation group interactions, density-vegetation group interactions and density-diver interactions were considered in the analysis of covariance. Three-way and higher level interaction effects were not considered because we

wished to derive a simple model with a relatively small number of parameters. Backward stepwise procedures were used to determine subsets of the six effects that explained the maximum amount of variability in the data with the smallest number of parameters. During the backward stepwise procedures, effects were included or eliminated from the model based on the probability level of F ratios for partial sums of squares.

Translation of predicted values from the logarithmic model, equation 13, back to the original scale, equation 12, required a correction for bias. The bias in the expected value of Y_{ijk} is $\exp(\frac{1}{2}\sigma^2)$ when the true variance of $Y_{ij}k$, σ^2, is known. Laurent (1963) gave an exact expression for the bias correction that incorporated additional terms when σ^2 was estimated from a sample. For the diver calibration data, the biases in estimating σ^2 from a sample were less than 0.05% (Biggs and Funk 1988), so expected values for Y_{ijk} were estimated from:

$$E(Y_{ijk}) = e^{a} e^{D_j} e^{V_k} X_{ijk}^{\beta_{jk}} e^{\frac{1}{2}s^2}; \qquad (14)$$

where,

s^2 = the mean squared error from the general linear model.

The variance of individual predicted Y_{ijk} was estimated from:

$$Var(Y_{ijk}) = [e^{(2Y_{ijk}+\sigma^2)}]\,[e^{\sigma^2}-1]. \qquad (15)$$

This expression is appropriate when σ^2 is known (Laurent 1963), but we used s^2 instead of σ^2 without correction for bias, since the bias introduced into estimates of the mean when s^2 was used for σ^2 were found to be small by Biggs and Funk (1988).

Spawning Biomass per Billion Eggs (B'). Data from the herring sampling program for AWL, sex ratio, and fecundity were used to estimate the relationship between spawning biomass and egg deposition. Once the age composition and sex ratio of a spawning population was determined, the average weight of the females in that population was calculated. The relationship between fecundity and female weight was used to calculate total numbers of eggs deposited and tonnes of herring spawners. The tonnes of spawning biomass required to produce one billion eggs (B') was estimated as:

$$B' = \frac{\bar{W}S}{F(\bar{W}_f)} 10^3; \qquad (16)$$

where,

$\bar{W}$ = estimated average weight in grams of all herring (male and female) in the spawning population in an area,

S = estimated ratio of total spawning biomass (male and female) to female spawning biomass,

$F(\bar{W}_f)$ = estimated fecundity at the average weight of females in the spawning population in an area, in numbers of eggs, and

$$10^3 = \text{conversion factor} = \frac{10^{-6}}{10^{-9}} = \frac{\text{grams to tonnes}}{\text{eggs to billions}}$$

Because average weight, sex ratio and fecundity were all estimated from the same herring samples, the estimates were not independent. The variance of B' is approximately:

$$\begin{aligned} Var(B') = (10^3)^2 \Big(& \left[\frac{S}{F(\bar{W}_f)}\right]^2 Var(\bar{W}) \\ & + \left[\frac{\bar{W}}{F(\bar{W}_f)}\right]^2 Var(S) \\ & + \left[\frac{\bar{W}S}{F(\bar{W}_f)^2}\right]^2 Var(F(\bar{W}_f f)) \\ & + 2Cov(\bar{W}, S) \left[\frac{S}{F(\bar{W}_f)}\right] \left[\frac{\bar{W}}{F(\bar{W}_f)}\right] \\ & - 2Cov[\bar{W}, F(\bar{W}_f)] \left[\frac{S}{F(\bar{W}_f)}\right] \left[\frac{\bar{W}S}{F(F(\bar{W}_f))^2}\right] \\ & - 2Cov[S, F(\bar{W}_f)] \left[\frac{\bar{W}}{F(\bar{W}_f)}\right] \left[\frac{\bar{W}S}{F(\bar{W}_f)^2}\right] \Big). \end{aligned} \quad (17)$$

Because S was estimated from pooled AWL samples, and in two of the areas S was estimated from a single AWL sample, it was not possible to estimate the covariance terms containing S, $Cov(\bar{W},S)$ and $Cov[S,F(\bar{W}_f)]$. These covariance terms were not included in the estimate of Var(B'), but probably contributed a negligible amount to Var(B'), because the term involving $Cov[\bar{W},F(\bar{W}_f)]$ was very small.

Herring Age, Weight, Length, Sex, and Fecundity

AWL information was collected from major concentrations of herring spawning in each of the five spawn deposition areas (Baker et al. 1991a; Baker et al. 1991b; Wilcock et al. In press). Because increased precision of spawn deposition biomass estimates was needed to evaluate oil spill impacts, AWL sampling in 1989-1991 was more intense than that regularly conducted by ADF&G. AWL sampling was also expanded to include collection of hydrocarbon, histopathological, and fecundity samples and to cover the cost of chartering vessels to collect herring samples as a direct result of the oil impact studies.

AWL Study Design and Sampling Procedures. Sampling generally began soon after concentrations of herring appeared in nearshore areas and were accessible to purse seines. Samples were taken from most large herring concentrations throughout PWS periodically over the duration of the spawning migration. AWL samples collected during the peak of spawning in each summary area, as determined from aerial survey sightings of milt and herring schools, were used

to estimate age and sex composition as well as average herring size from all major biomass concentrations in each area.

AWL sampling was stratified by date and area for each commercial fishery and for test fishing catches in each spawning area. Sample size for each stratum was set to simultaneously estimate proportions by age when sampling from a multinomial population (Thompson 1987). The goal was to select the smallest sample size for a random sample from a multinomial population such that the probability was at least $1-\alpha$ (precision = 0.05) that all the estimated proportions were simultaneously within 5% (accuracy = 0.05) of the true population age proportions. A sample size of 450 herring per stratum was set to ensure that this level of precision and accuracy would be obtained for any number of age classes and proportions when less than 5% of the collected scales were unreadable. Wilcock et. al. (*In press*) provide a thorough description of PWS herring AWL sampling program procedures.

Fecundity Estimation. Subsamples of female herring, selected from AWL samples collected during 1989-1992 and stratified by summary area (Figure 1), were used to estimate egg and gonad weight. Egg and gonad weights were used to calculate average fecundity at the average female weight ($F(\bar{W}_f)$) from expression (16).

A fecundity sampling goal was set based upon the precision of the biomass estimate. Since the spawn deposition survey attempted to estimate spawning biomass with 95% confidence intervals of no more than ± 25% of the biomass estimate, we wanted the fecundity estimate to contribute no more than 1% to confidence interval width. It was determined that approximately 100 female herring of exactly the mean weight of the female herring in the spawning population were needed. Because mean female weight was unknown at the time of sampling, more females had to be sampled over a range of sizes and from all areas. Mean weight was predicted from annual forecasts and samples were stratified by length such that a larger proportion of females would be sampled around the predicted mean weight of females in the population. Fecundity samples were collected such that approximately 100 females were collected from each area. This sampling procedure not only met precision levels for the biomass estimate but allowed for comparisons of fecundity both temporally (between years) and spatially (between areas).

Ovaries were removed from each female herring sampled, placed in ziploc bags, frozen, and processed at a later date. Sample location, date and AWL identification number were also recorded. After thawing, each ovary was weighed to the nearest 0.01 g. In 1988, prior to NRDA studies, only one subsample of 0.5 to 1.0 g of eggs was removed from each ovary collected in 1988 and soaked in Gilson's fluid (Nielsen and Johnson 1987) until the eggs were opaque and loose from the connective membrane. Eggs were then counted under magnification and total fecundity was estimated using gravimetric expansion. In 1989-1991, during NRDA studies, four 0.1 to 0.3 g (approximately 200 eggs) subsamples were randomly taken from each roe sample, placed in labeled petri dishes, and weighed. Gilson's fluid was then poured into the petri dishes and allowed to soak for a minimum of 5 minutes to loosen the eggs from the connective membrane. Once the eggs were loose, the Gilson's fluid was decanted and the number of eggs in each subsample were counted. Fecundity per female was estimated by multiplying the mean number of eggs per gram in the subsamples by the weight of the ovaries. The gonad weight was assumed to be the equivalent of the weight of the ovaries removed from each female. Gonosomatic index (GSI) was defined as the percentage of total herring weight accounted for by gonad weight and was calculated by dividing the gonad weight by body weight of each female sampled.

Based on results of a pilot study in 1991, a more accurate method of estimating fecundity, in which ovaries were preserved in Gilson's solution rather than being frozen, was adopted for use in 1992:

1) Sample labels were removed from inside the jars containing ovaries and the AWL sample number, date, and collection location were recorded on a data form.

2) The sample bottle was vigorously shaken to separate eggs and liberate them from the connective membrane. The sample was poured into a fine mesh sieve and the jar was rinsed to retrieve any remaining eggs. The sample was rinsed with water for a few minutes to wash away Gilson's Solution and dissolved connective membrane. All possible membrane was separated from the sample, weighed, and recorded on the data form.

4) Clumps of eggs were carefully broken apart with a spatula. Most samples had a rubbery consistency and separated easily. If eggs were unusually soft, handling was kept to a minimum to avoid crushing any eggs.

5) A vacuum pump and filter funnel system were used to extract water from the eggs. A wet filter was placed inside the filter funnel. Eggs were carefully transferred from the sieve to the vacuum pump. The vacuum pump was run until water ceased to drip from the funnel. The time needed to remove the water (generally about two minutes) varied depending on the size of the sample and the amount of membrane. If membrane clogged the filter, the sample was rinsed in the sieve a second time, and vacuum filtered again.

6) The sample was removed from the funnel and any visible membrane was separated. Membrane and eggs were weighed separately and recorded. Five subsamples of about 0.10 to 0.30 g (100-200 eggs) were randomly selected from the sample and placed into separate petri dishes. The number of eggs in each subsample was counted with a tally counter and recorded on the data form.

AWL Data Analysis

Mean Weight and Sex Ratio. Mean weight and sex ratio were estimated from AWL samples collected from each of the five spawn deposition areas. AWL samples collected during peak spawning in each area were pooled to estimate mean weight and sex ratio for that area. Average weight and sex ratio for PWS were estimated as a weighted average of estimates from each area. Average weight and sex ratio for each area were weighted by the escapement biomass estimate based on spawn deposition surveys for that area.

Sex ratio, S, was calculated as the ratio of the number of herring of both sexes in AWL samples to the number of females. The binomial distribution is applicable to estimating the proportion, p, of females in AWL samples, where S = 1/p. The variance of S is then given by:

$$Var(S) = \frac{S^2(S-1)}{n}, \qquad (18)$$

where n is the number of fish in the AWL sample.

Fecundity for Biomass Estimates. Average fecundity for each area was estimated from the fecundity-weight relationship, using average female weight in the AWL samples from each area, and applied to the spawn deposition biomass estimator ($F(\bar{W}_f)$ in equation 16). The variance of estimated average fecundities was approximated by the variance of predicted means from the fecundity-weight linear regression (Draper and Smith 1981):

$$Var[F(\bar{W}_f)] = s^2\left[\frac{1}{n}+\frac{1}{q}+\frac{(\bar{W}_f-\overline{WF})^2}{\sum (W_i-\overline{WF})^2}\right] \qquad (19)$$

where

s^2 = the residual mean square from the fecundity-weight linear regression,
$\bar{W}_f$ = the average weight of female fish in the spawning population,
$\overline{WF}$ = the average weight of females in the fecundity sample,
W_i = the weights of individual females in the fecundity sample,
n = the total number of females in the fecundity sample from each area,
q = the total number of females in the representative AWL sample or pooled samples from the corresponding area.

Fecundity for Damage Assessment. The first objective of this study component was to estimate the fecundity of all major concentrations of herring in PWS. Hourston et al. (1981) found that female body weight at spawning explained 70% of the variation in fecundity among individuals while length and age only explained another 2% of the variation. Therefore, we used body weight as the dependent variable to estimate fecundity. In addition, linear relationships between female body weight and fecundity, egg weight, and gonad weight were developed. Year and area differences were examined using analysis of covariance (ANCOVA) models that included all possible interaction terms:

$$\log(Y_{ijk}) = \mu + \log(wt_i) + year_j + area_k + (\log(wt))(year_{ij}) + (\log(wt))(area_{ik}) + (year)(area_{jk}) + (\log(wt))(year)(area_{ijk}) + \epsilon_{ijk} \qquad (20)$$

where

Y_{ijk} = fecundity, egg size, or gonad weight of the ith female herring in the jth year and kth area,
μ = grand mean,
$year_j$ = fixed effect of the jth year,
$area_k$ = the fixed effect of the kth area, and
ϵ_{jk} = error term.

The second objective of this study component was to determine the effects of the oil spill on the reproductive potential of female herring. Comparison of fecundity, egg size, and gonad weight, by year and area, was based on a full two way analysis of variance (ANOVA) model:

$$\log(Y_{ijk}) = \mu + year_j + area_k + (year)(area_{jk}) + \epsilon_{jk} \quad (21)$$

The covariate body weight was not used in this comparison since measurements of body weight were collected along with fecundity data and may have been influenced by the oil spill.

Dependent variables were log transformed for all ANCOVA and ANOVA models to help stabilize the variance. All terms with probabilities of occurrence less than 0.05 were considered significant. The SAS general linear model (GLM) procedure was used to fit the ANCOVA and ANOVA models (SAS 1987). To examine the data for oiling effects upon fecundity, the following assumptions were made: 1) Montague and Naked Island areas were contaminated ("oiled"), while North, Northeast, and Southeast shore areas were be uncontaminated ("unoiled"); 2) data from 1988 and 1989 represented uncontaminated ("pre-spill") conditions, while those collected in 1990, 1991, and 1992 represented contaminated ("post-spill") conditions. Contrast statements in the GLM procedures of SAS were used to make comparisons (SAS 1987). If the oil spill affected fecundity, we would expect to find, for example, no difference between "oiled" and "unoiled" sites during pre-spill years, but a significant difference during post-spill years (probably diminishing each year).

We were also concerned that natural fluctuations in reproductive rates would confound or mask oil spill effects. Sea surface temperature appears to be an important natural factor affecting reproductive potential of herring. Tanasichuk and Ware (1987) found that sea surface temperatures 60 to 90 days before spawning best accounted for variations in size specific fecundity for herring in British Columbia, Canada. While no sea surface temperature data were collected for this study, sea surface temperature anomaly (SSTA) data were available from 1947-1992 for three grid points (55°N, 160°W; 60°N, 150°W; 60°N, 155°W) in the western Gulf of Alaska that were in close proximity to PWS (Figure 2). These data included mean seasonal SSTA for three month periods (Jan, Feb, Mar; Apr, May, Jun; Jul, Aug, Sep; and Oct, Nov, Dec) from 1947-1992. SSTA data were originally collected by ships and more recently by satellite infrared sensors. Data for our analyses were compiled by Scripps Institute of Oceanography in LaJolla, California, and provided to us by Mark Willette (Alaska Department of Fish and Game, personal communications). We use mean seasonal SSTA data 13-15, 10-12, 7-9, 4-6, and 1-3 months prior to and during spawning, for the years 1987-1992, as independent variables in linear regressions in which mean body weight, fecundity, gonad weight, GSI, or egg weight for all female herring collected in PWS were used as independent variables.

Egg Loss Study

Published estimates of herring egg loss between the time of spawning and the time of egg deposition were relatively high before diver surveys were extensively used to improve the accuracy of these estimates. For example, Montgomery (1958) estimated that egg loss in Southeast Alaska ranged from 25% to 40%. Similar egg losses were used in early studies of egg deposition in Southeast Alaska described by Blankenbeckler and Larson (1987). However, Haegele et al. (1981) argued that these estimates too high since most spawning was actually subtidal rather than intertidal and predation and wave loss was probably less for subtidal spawn. Haegele et al. (1981) estimated egg loss to be approximately 10% from predation and from wave action loosening the eggs from the substrate during storms. Since the timing of diver surveys following spawning is similar in British Columbia, Southeastern Alaska, and PWS, the 10% egg loss used in British Columbia and Southeast Alaska (W. Blankenbeckler, Alaska Department of Fish and Game, Ketchikan, personal communication) was

used for the PWS egg deposition survey. While methods and results from the egg loss study conducted in 1990-1991 in PWS were included in this report, analyses were not completed in time to incorporate them in spawn biomass estimates.

Nine egg loss transects were established in 1990 and ten transects in 1991 after herring had first spawned in PWS (Figure 3a and b). Transects were chosen to represent major spawning areas and the range of depths where herring usually spawn (+1.65 m to -9.90 m). Transects were established perpendicular to shore following a compass course. For PWS, Rosenthal (1976) found that most herring spawn occurred in a area extending from the intertidal zone to -11.55 m in depth. More recently, Funk et al.(*in prep*) found that herring spawn was rarely deposited below a depth of -9.90 m. Therefore, sampling stations were set at six depth levels: 1.65 m (5 ft), 0.33 m (1 ft), 0.00 m (0 ft), -1.65 m (-5 ft), -4.95 m (-15 ft), and -9.90 m (-30 ft). Station depths were located using SCUBA diver depth gauges, and readings were later corrected for tide level.

Sampling methods differed between 1990 and 1991. In 1990, a marked leadline placed perpendicular to the transect and parallel to the shoreline was used to sample at each depth level along a transect. The leadline contained five marks along it, and the number of eggs was estimated within a 0.1 m^2 sample quadrat placed at each mark as well as at the end of the leadline. This sixth quadrat sample was used as a calibration sample, and all eggs and vegetation were removed and later counted in the Cordova ADF&G laboratory following procedures previously explained for spawn deposition samples. Five egg estimates were made at each of the six depth levels, and egg estimates were made every other day from the time of spawning in each area until the time of hatching (a period of approximately 24-30 days). In 1991, a marked leadline was not used. Instead, a 5 x 2 grid of permanent 0.1 m^2 quadrats was placed at each transect and depth. Each grid contained 10 - 0.1 m^2 quadrats that were placed perpendicular to the transects and parallel to the shoreline. Permanent grids allowed divers to estimate the number of egg in the same grid over time. Divers made five estimates of egg density within the 0.1 m^2 quadrat grid. Finally, divers placed a 0.1 m^2 quadrat at the en0d of the grid, made a sixth estimate of egg density, and collected all the eggs and vegetation within the quadrat. As in 1990, counts of eggs were made in the Cordova ADF&G laboratory at a later time. Eggs were counted every three to four days in 1991.

Systematic biases among divers and among vegetation types were found to exist in diver estimates from previous spawn deposition surveys in Prince William Sound (Funk et al. *in prep*). We assumed the same biases would exist in our diver estimates, and accounted for them as we had in spawn deposition surveys (Funk et al. *in prep*). Diver estimates of egg density were approximately proportional to laboratory-enumerated counts. The model used to account for biases in diver estimates was identical to the model described by equations 12 and 13.

After adjusting egg estimates, we selected an exponential decay model to estimate loss in numbers of eggs over time:

$$ADJ_{ijk} = e^{\alpha} e^{trans_j} e^{depth_k} e^{\tau_{jk}(days_{ijk})} e^{\varepsilon} \qquad (22)$$

where,

α = a constant,
ADJ_{ijk} = adjusted egg density estimates,
$trans_j$ = parameters representing the effect of transect j,

$depth_k$	=	parameters representing the effect of depth k,
τ_{jk}	=	parameters controlling the functional form of the relationship between egg density and time (number of days after spawning),
$days_{ijk}$	=	the number of days after spawning occurred, and
ε	=	normally distributed random variable with mean = 0 and variance = σ^2.

A multiplicative model was chosen because egg numbers were expected to vary with location (transect) and depth. All interactive terms were included in the model. After a logarithmic transformation, equation 4 became:

$$\log_e(ADJ_{ijk}) = \alpha + trans_j + depth_k + \tau_{jk}(days_{ijk}) + \epsilon \qquad (23)$$

In logarithmic form, the model comprised a linear analysis of covariance (ANCOVA) with two factor effects (transect and depth) and 1 covariate (number of days after spawning). SAS (1987) procedure for general linear models (GLM) was used to obtain least squares estimates of the parameters. Estimates of eggs over time (days) were then made for each transect and depth.

Hydrocarbon Sampling and Analysis

Hydrocarbon samples were collected from herring eggs, adult herring tissues, and whole mussels in 1989, 1990, and 1991 from oiled and unoiled areas. Samples collected in 1989 and 1990 were tested, but results were not available until October, 1992. Samples from 1991 were not processed. Only results of mussel tissue testing were presented in this report because time and money were not available to perform principal component analysis to index other samples (Jeff Short, NOAA-NMFS, personal communication). Mussel tissue results were used to complete herring injury assessment analyses, also oil levels in herring eggs and adult tissues are also reported.

Mussel Tissue Samples

Mussel tissue samples were collected at each study site used exclusively for injury assessment. These sites did not overlap sites used for spawn deposition surveys. Mussels were collected from 15 oiled sites and 5 unoiled sites in 1989 (Figure 3c). Mussels were collected from 5 previously oiled sites and 3 unoiled sites in 1990. To ensure proximity to and overlap with herring embryo distribution, mussels were collected at the lowest depth possible, about 0 to +1 Mean Low Low Water (MLLW). Mussel tissues were analyzed for the presence and level of petroleum hydrocarbons to document exposure to oil spilled by the *T/V Exxon Valdez*. Analyses of these samples for petroleum hydrocarbons and metabolites of petroleum hydrocarbons for Fish/Shellfish Study 11, and all other NRDA studies, were coordinated by NOAA and United States Fish and Wildlife Service (USFWS) as part of NRDA Technical Services (TS) Study 1 - Hydrocarbon Analytical Services and Analysis of Distribution and Weathering of Spilled Oil (Manen 1993). Each tissue sample was analyzed for 63 independent analytes using mass spectrometry and gas chromatography (MS/GC). Analytes levels were reported in ng/g wet weight or ppb. The resulting hydrocarbon data set was evaluated at NMFS, Auke Bay Laboratory for internal consistency and to identify the presence of *T/V Exxon Valdez* petroleum hydrocarbons (Short and Heintz *In prep*). Inconsistent hydrocarbons were identified using computer-based statistical methods to identify groups of samples that were biased systematically, or that were

exposed to extraneous contamination unrelated to the oil spill. These data were removed from the hydrocarbon data set.

A reliable quantitative measure of petroleum hydrocarbon contamination was sought to compare the levels of hydrocarbons present at sampling locations and to determine if the source of contamination was oil from the *T/V Exxon Valdez*. This measure also needed to be comparable with previous studies reported in the scientific literature. Total aromatic hydrocarbons (the sum of selected aromatic hydrocarbon groups) and phytane were selected to be used as indicators in this study. In general, presence of aromatic hydrocarbons and phytane are evidence of the presence of oil. In addition, historic, 1977-1980, levels of aromatic hydrocarbons and phytane in PWS were measured and reported by Karinen et. al. (1993). Babcock and Karinen (1991) reported relative levels of phenanthrene, dibenzothiophenes, fluorenes, chrysenes, and phytane which are indicative of Prudhoe Bay Crude Oil (PBCO). We assumed that levels of aromatic hydrocarbons and phytane in PBCO was approximately equivalent to the oil spilled by the T/V *Exxon Valdez* (Figure 4). In addition, Short and Heintz (*In prep*) developed a single oil concentration based on principal component analysis of the hydrocarbon data set. This concentration was an estimate of hydrocarbons present in mussel tissues resulting from the *Exxon Valdez* oil spill.

RESULTS

Exposure of Herring to Oil

On March 24, 1989, crude oil leaching from the grounded *T/V Exxon Valdez* began forming a large pool to the south and west of the vessel. This pool remained in place, spreading over several miles by March 26, day 3 of the spill (Figure 5). By day 4, surface oil had moved to the southwest and had begun to surround the Naked Island group (Figure 6). By day 5 surface oil was beginning to exit PWS through Hinchinbrook Entrance, but several sheens had broken loose from the main oil mass (Figure 7). By day 11, April 4, oil was still moving steadily out of PWS (Figure 8) and the composite trajectory had covered much of the area traditionally utilized by spawning adult herring (Figure 9). Fate, bioavailability, and toxicity of the oil in surface and subsurface waters were not known until three years after the spill.

On March 30, 1989, the first schools of herring staging for spawning were sighted in northeast PWS near the village of Tatitlek (Table 1). By April 1, large numbers of herring had been sighted in the Northeast area and in the North Shore area near Granite Point and Fairmont Bay. Both groups of herring faced a high probability of encountering oil during their migration to the spawning grounds, although the Northeast and North Shore areas were not oiled. By April 12, the number of herring observed from aerial surveys peaked in areas within the oil trajectory: Naked Island group, Montague Island, and Green Island (Table 1).

During peak herring spawning in April 1989, the composite oil trajectory overlapped 43.0% of the total shoreline used for spawning in 1989 (Figure 9; Table 2 and 3). Fifty-two percent of the herring biomass spawned in oiled areas, and an even greater percentage of the spawning population could have encountered oil during their migration. Additionally, based on historical fishing areas documented from 1914 to the present, about 80% of areas used by herring for summer rearing and feeding overlapped the oil trajectory (Reid 1971; Rounsefell 1931). In 1990, 31.4% of the shoreline used for herring spawning and 40.1% of the spawning biomass was documented to occur in

previously oiled areas (Table 4). Hydrocarbon levels were still elevated at some of the oiled sites, but in general were much lower in 1990 than in 1989. In 1991, 42.1% of the shoreline used for spawning and 74.0% of the spawning biomass occurred in previously oiled areas (Table 5). In 1992, 47.3% of the shoreline used for spawning and 50.1% of the spawning biomass occurred in previously oiled areas (Table 6).

Over the course of the four-year period from 1989 to 1992, 30 sites were established in oiled areas and 21 in control areas to meet the various objectives of the study (Figure 3a). Of all the oiled areas with herring spawn, only the western shores of Zaikof Bay on northern Montague Island escaped exposure to floating oil. By mid-April in 1989 when herring spawning activity was at its peak, tar balls or other visible signs of crude oil was observed at only a few sites. Tar balls were found in the upper intertidal area of Rocky Bay sites O17-19 and sites O25-27 on the west coast of north end of Montague Island. Site O3 in Bass Harbor of Naked Island was inside an embayment that was closed off by an oil boom. Floating oil was being stored for skimming operations behind this boom when eggs where incubating there. Tar balls were also visible in the upper intertidal above sites O13, O14 and O23 near or in Cabin Bay on Naked Island. Oil was not observed at any of the control sites.

The Prince William Sound Herring Population

Biomass estimates from spawn deposition surveys were calculated separately for each of the five major areas each year, providing annual estimates of the PWS herring spawning population (Tables 3-6). The number of diver survey transects without eggs, identified during skiff surveys used to correct spawn maps based on aerial surveys, decreased from 21% of the total in 1988 to 13% in 1989. Since skiff surveys were time consuming and did not significantly improve the accuracy of overall biomass estimates, they were discontinued after 1990. Instead, divers surveyed edges of sightings mapped from the air to determine the limits of actual egg deposition. Using the dive team to determine the actual extent of egg deposition was quick, easy, and as effective as skiff surveys.

Run Timing and Spawning Activity

In 1989, a total of 98.4 miles of spawn was mapped during aerial surveys completed between March 30 and April 27 (Figure 7; Tables 1, 2 and 3). Aerial surveys were conducted after April 27, but spawning activity had ceased and biomass estimates dropped to zero. Peak herring spawning biomass occurred on April 12 (Table 1). This was seven days earlier than the historic, 1973-1992, average date of April 19 calculated from a mean timing curve (Biggs et al. 1993; Table 7; Figure 11). The first spawning activity observed during aerial surveys occurred near Tatitlek on March 31 (Table 2). By April 13, spawning had peaked in the Northeast, North Shore, and Naked Island areas. Five days later, April 18, spawning activity peaked at Montague and Green Islands. The mean date of peak spawning activity for PWS in 1989 was April 13; eight days earlier than the 1973 to 1992 average, April 21, calculated from a mean spawn timing curve. Although the timing of spawning was early, the distribution of the spawn by area appeared normal compared to previous years and did not seem to be affected by the oil spill.

In 1990, a total of 94.1 miles of spawn was mapped during aerial and skiff surveys completed between March 31 and April 23 (Figure 10; Table 4; Biggs et al. 1993a). Peak herring spawning biomass occurred on April 13. As occurred in 1989, this was seven days earlier than the historic, 1973-1992, average

date of April 19. The first spawning activity observed during aerial surveys occurred in Port Gravina on April 1. Spawning peaked in the Northeast area by April 14 and in the North Shore, Naked Island, and Montague Island (including Green Island) areas by April 16. The mean date of peak spawning activity for PWS in 1990 was April 16; five days earlier than the historic, 1973-1992, average. Spawning in both 1989 and 1990 was the earliest recorded since 1973 (Figure 11; Table 7). In addition, spawning throughout PWS occurred over a shorter period of time than in past years. Distribution of the spawn changed slightly from 1989 with less spawning in the North Shore and Naked Island areas, and more spawn in the Northeast area (Figure 12).

The 1991 season was characterized by a cold winter, late spring and continuous storm events throughout the spawning season. A total of 58.0 miles of spawn was mapped during aerial surveys completed between March 30 and May 12 (Figure 13; Table 5; Biggs et al. 1993). Peak herring spawning biomass occurred on April 15, four days earlier than the historic, 1973-1992, average date of April 19. The first spawning activity observed during aerial surveys occurred in Port Gravina on April 1. Spawning peaked in the Northeast area by April 14 and in the North Shore, Naked Island, and Montague Island (including Green Island) areas by April 16. The mean date of peak spawning activity for PWS in 1991 was April 22; one day later than the historic, 1973-1992, average (Table 7). Probably due to colder than average water temperatures, herring remained near spawning areas for a longer period of time prior to spawning than in previous years. Distribution of the spawn changed slightly, and spawning areas in the North Shore and Naked Island areas were smaller in 1991 than in 1990 and 1989 (Figure 12).

In 1992, water temperatures and weather patterns returned to more normal conditions. A total of 74.7 miles of spawn was mapped during aerial surveys completed between March 30 and April 27 (Figure 14; Table 6; Biggs et al. 1993). Peak herring spawning biomass occurred on April 15, four days earlier than the historic, 1973-1992, average date of April 19 (Table 7). The first spawning activity observed during aerial surveys occurred in Port Gravina, Tatitlek Narrows, and Hinchinbrook Island on April 7 (Figure 14). Spawning peaked in the Southeast area by April 14, and in the Northeast area by April 17. No spawning was observed in the North Shore or Green Island areas in 1992, and only 0.3 miles of spawn were recorded in late in the season, April 26 and 27, in Naked Island area. Most spawning peaked by April 24, when a total of 35.0 miles was recorded. The mean date of peak spawning activity for PWS in 1992 was April 21, the same date as the historic, 1973-1992, average date (Table 7). Distribution of spawn was similar to 1991: no spawn occurred in North Shore area and less than 1% of the total was deposited at Naked Island (Table 6; Figure 12).

In order to understand the relationship between run timing and environmental factors affecting it, an analysis was conducted. An ANOVA between the mean date of herring spawning in PWS and winter and spring Gulf of Alaska sea-surface temperature anomalies was significant($P<0.05$). The relationship between mean spawning date and either winter or spring sea-surface temperature anomalies, tested separately, was not significant. All three parameters were normally distributed using a Shapiro-Wilk test (SAS 1987) and there the ANOVA test was found to be appropriate. Winter temperatures were positively correlated with run timing with colder sea-surface water inducing early run timing and the opposite true for spring temperatures (negatively correlated). The combination of cold winters and warm springs seemed to produce the earliest run timing.

Biomass Estimates

The 1989 herring escapement biomass, based on the spawn survey, was estimated to be 45,281 tonnes with a 95% confidence interval of ± 22% (Table 3). This was the smallest confidence interval error ever calculated for an escapement biomass estimate (the confidence interval for the 1988 estimate was ± 29.5%). The 1989 escapement biomass estimate was within 9% of the preseason run biomass forecast of 49,794.5 tonnes based on the 1988 spawn deposition survey (Brannian 1989). Commercial spring herring fisheries were originally allocated a catch of 8,337.3 tonnes, but these fisheries were closed in 1989 to avoid product contamination by oil. The greater than expected escapement effectively increased the biomass of spawning herring by 16.0% in 1989. Spawning biomass was evenly distributed between oiled and unoiled areas according to the egg survey results.

The 1990 herring escapement biomass, based on the spawn survey, was estimated to be 115,646 tonnes with a 95% confidence interval of ± 27%, which was slightly greater than the accuracy goal of 25% (Table 4). The escapement biomass estimate was more than twice preseason run biomass forecast of 46,894.1 tonnes (Baker 1990). Actual estimated run biomass, 125,699 tonnes, was 2.7 times greater than the preseason forecast when the commercial fishery catch of 10,053.2 tonnes was added to the escapement biomass.

The 1991 herring escapement biomass, based on the spawn survey, was estimated to be 127,881 tonnes with a 95% confidence interval of ± 32% (Table 5). Although only a few years of estimates were available, accuracy of the spawn deposition estimate seemed to decline with increasing biomass (Figure 15). The escapement biomass estimate was about 1.5 times greater than the preseason run biomass forecast of 87,693.8 tonnes (Baker 1991). Actual estimated run biomass, 142,447 tonnes, was 1.6 times greater than the preseason forecast when the commercial fishery catch of 14,565.6 tonnes was added to the escapement biomass.

The 1992 herring escapement biomass, based on the spawn survey, was estimated to be 116,358 tonnes with a 95% confidence interval of ± 25% (Figure 15; Table 6). The escapement biomass estimate was within 6% of the preseason run biomass forecast of 110,080 tonnes (Biggs and Baker 1992). Actual estimated run biomass, 137,015.3 tonnes, was still only 25% greater than the preseason forecast when the record commercial catch of 20,657.3 tonnes was added to the escapement biomass.

In collecting egg abundance data for the spawn deposition survey, used mainly to back-calculate escapement biomass, other detailed information about egg density, spawn widths, spawn substrate, etc. are obtained. It is known that egg density can affect egg hatching success and survival obscuring oil spill effects on those parameters. In PWS, egg density and spawn width both increased from 1988 until 1991 and then decreased in 1992 (Figure 16). Since egg density was lower in 1989 than in subsequent years after the spill, egg mortality due to overcrowding should not have confounded oiling effects during the spill year.

Escapement biomass estimates based on spawn surveys made from 1988 to 1992 were compared to other indices of spawning biomass available for 1972 to 1992 (Figure 17). Age-structured-analysis (ASA) of PWS herring data has produced estimates which generally fall close to or within the bounds of spawn survey 95% confidence intervals (Fritz Funk, Alaska Department of Fish and Game, personal communication). However, the ASA estimate for 1989 was about 2.4 times greater than the spawn survey estimate for that year, and almost 2 times greater than the upper 95% confidence limit. Estimates of shoreline miles of milt based on aerial surveys generally agree more closely with ASA model estimates than with spawn survey estimates.

Fecundity, Egg Size, and Gonadal Weight

The number of female herring sampled for fecundity ranged from 218 in 1991 to 415 in 1989 (Table 8). Differences in sample sizes among years occurred because herring did not spawn in all five areas each year or because efforts to sample small schools of spawners failed. Fecundity samples were collected from the Montague Island and Northeast areas all five years; the North shore area during 1988-1991; the Naked Island area in 1989, 1990, and 1992; and the Southeast area in 1992.

Mean body weight of female herring sampled for fecundity was 129.0 g, while individual values ranged from 107.8 g at Montague Island in 1988 to 152.0 g in the North Shore area in 1992 (Table 8). Mean fecundity was 18,992 eggs, while individual values ranged from 15,175 eggs at Montague Island in 1988 to 22,129 eggs in the North Shore area in 1992 (Table 9). Mean egg weight was 0.00146 g, while individual values ranged from 0.00120 g in the Northeast Shore area in 1992 to 0.0018 g at Naked Island in 1990 (Table 10). Mean gonad weight was 27.4 g, while individual values ranged from 21.5 g in the Southeast Shore area in 1992 to 36.6 g in the Northeast Shore area in 1990 (Table 11). Mean GSI was 21%, while individual values ranged from 18% in the Southeast Shore area in 1992 to 23% in the Northeast Shore area in 1990 (Table 12).

Fecundity, egg weight, and gonad weight all tended to increase with increasing body weight in all areas and for all years examined (Figures 18-23). GSI showed a similar relationship to body weight in all areas and for all years examined (Figures 24-25). ANCOVA models, based on logarithmic transformed data, confirmed that herring fecundity, egg weight, and gonad weight varied significantly ($P<0.0001$) with body weight, area, and year. The most significant ($P<0.0001$) term in all ANCOVA models was body weight, which accounted for 66.7%, 32.7%, and 81.8% of the variability observed for fecundity, egg weight, and gonad weight data, respectively. Several interactive effects (i.e. year-area; weight-year; weight-area; weight-year-area) were also significant in the models, suggesting that intercepts and slopes of linear relationships were different among years and areas.

ANOVA models, based on logarithmic transformed data, showed that area and year effects explained a significant ($P<0,001$) portion of the variability observed for fecundity, egg weight and gonad weight (Table 13). These two factors accounted for 12.4%, 22.6%, and 16.7% of the variability observed for fecundity, egg weight, and gonad weight data, respectively. Year-area interactions were significant ($P<0.001$) in all three models. Contrasts which examined differences between the Northeast area mean and the other area means in 1990 to those in 1989 were significant for fecundity ($P<0.0001$), egg weight ($P<0.0011$), and gonad weight ($P<0.0055$). Examination of year-area interaction least square means showed that values for fecundity, egg weight, and gonad weight were generally greater in 1990, the year after the spill, than in 1989, the spill year, although differences existed among areas (Figure 26). Egg and gonad weight increased in all four areas between 1989 and 1990, but fecundity increased only in the Northeast Shore area, while the other four areas showed no change.

SSTA's generally explained a large portion of the variation observed in body weight, gonad weight, GSI, fecundity, and egg weight only prior to spawning (Table 14, Figure 27). Greatest values for the coefficient of determination (R^2) were found 13 to 15 months prior to spawning for body weight (0.827), gonad weight (0.777) and fecundity (0.657), and 7 to 9 months prior to spawning for GSI (0.808) and egg weight (0.909). Temperatures 13 to 15 months prior to spawning were warmer than average in 1988 and 1989 and colder than normal during 1990-1992 (Figure 28). Temperatures 7 to 9 months prior to

spawning were warmer than average during 1988-1991 and slightly colder than average in 1992. SSTA's 4 to 6 months prior to spawning were warmer than average in 1988, 1989, 1991 and 1992 and colder than average in 1990. Body weight, gonad weight, GSI, and fecundity were generally lowest when temperatures 13 to 15 months prior to spawning were warmer than average (1988 and 1989), and highest when temperatures 13 to 15 months prior to spawning were colder than average (1990 and 1991). Body weight, gonad weight, GSI, and egg weight generally increased as temperatures 7 to 9 months prior to spawning increased. Egg weight was greatest when temperatures 4 to 6 months prior to spawning were coldest (1990) and least when temperatures 4 to 6 months prior to spawning were warmest (1992).

Egg Loss Study

During the two years of the egg loss study, 1990 and 1991, herring were first observed spawning between April 15 and 25 April (Table 15). Based upon observations at egg loss transect sampling locations, the mean number of days from the time spawning was first observed to the time the first eyed eggs were seen was 17.9 days in 1990 and 17.4 days in 1991. Mean number of days from the time of spawning was first observed to the time eggs began to hatch was 23.6 days in 1990 and 23.4 days in 1991 (Table 15; Figure 29).

Egg estimates by SCUBA divers were calibrated against laboratory counts for the sixth egg sample at each depth. The simplest model found to adjust diver estimates, which also accounted for differences related to vegetation type, took the following form:

$$\log(ADJ_{ijk}) = D_j + V_k + V_k \log(X_{ik}) \qquad (24)$$

The model was highly significant ($P<0.0001$) and explained 82.3% of the variability in the data (Table 16).

After correcting diver estimates using equation 1328, equation 23 was fit to egg estimates for 1990 and 1991. ANCOVA models were significant ($P<0.0001$) for both years and accounted for 65.3% of the variability in egg estimates in 1990 and 76.3% in 1991 (Tables 17 and 18). All effects and their interactions were significant ($P<0.05$). Chosen models were then used to estimate the relative percent loss or gain in eggs for all transects, depths, and kelp types in 1990 (Figures 30-32) and 1991 (Figures 33-34).

Changes in egg density by depth at each transect in 1990 and 1991 were highly variable, and in some cases estimated density actually increased over time (Figure 30-34). However, since we were interested in estimating overall egg loss for all of PWS, we calculated mean daily changes in egg density by depth for each major spawning area in 1990 and 1991 (Table 19, Figure 35). In most areas, egg loss was greatest in the intertidal zone at 1.65 m (Figure 35). Egg loss at all other depth levels, 0.33 m to -9.90 m, were similar. In 1990, the North Shore area had a daily mean gain of 0.2% at all depths, while Montague and Naked Islands had daily mean losses of -9.4% and -7.0%, respectively. In 1991, Montague Island had a daily mean gain of 1.8%, while the North Shore and Northeast areas had daily mean losses of -3.5% and -4.7%.

For egg loss transects, the mean number of days between the date herring began to spawn and the date a spawn deposition survey would have been conducted (i.e. five days after spawning ended) was 7 in 1990 and 10 in 1991. Using daily mean egg loss estimates calculated from data for all areas and depths for each year (-5.4% in 1990, -2.1% in 1991; Table 19), 37.8% of all eggs

would have been lost prior to surveys in 1990 and 21.0% in 1991. Using daily mean egg loss estimates calculated without data from the 1.65 m level (-4.0% in 1990, -0.2% in 1991), 28.0% of all eggs would have been lost prior to surveys in 1990 and 2.0% in 1991. Using daily mean egg loss estimates for all areas and both years, but without data from the 1.65 m level (-2.1%), 14.7% of all eggs would have been lost prior to surveys in 1990 and 21.0% in 1991.

An estimate of total egg mortality would be given by calculating total egg loss from time of spawning to time of hatching. The mean number of days from spawning to hatching was almost 24 in both 1990 and 1991. Using daily mean egg loss estimates for all areas and both years, 91.2% of all eggs would have been lost prior to hatching, if data from the 1.65 m depth level were included in calculating the mean (-3.8% d^{-1}), and 50.4% of all eggs would have been lost, if data from the 1.65 m level were excluded (-2.1% d^{-1}).

Hydrocarbon Sampling and Analysis

In 1989, a total of 75 mussel tissue samples were collected and analyzed for the presence of hydrocarbons: 60 samples from oiled sites and 15 samples from unoiled sites (Figures 3a and b; Table 20). Triplicate samples were obtained from each of 19 oiled sites (including samples from two depths at one oiled site) and 5 unoiled sites.

In 1990, a total of 33 mussel tissue samples were collected and analyzed for the presence of hydrocarbons: 24 samples from oiled sites and 9 samples from unoiled sites (Table 21). Triplicate samples were obtained from each of six oiled sites and each of three unoiled sites.

In 1989, mean levels of aromatic hydrocarbons and phytane were low in mussels from unoiled sites, and much higher, and more variable, in mussels from oiled sites (Table 20; Figure 36). Total aromatic hydrocarbon concentrations ranged from 37.4 to 141.3 ng/g wet weight (ppb) at unoiled sites, and from 115.2 to 2,532.8 ng/g wet weight (ppb) at oiled sites. Phytane ranged from 0.0 to 3.1 ng/g wet weight (ppb) at unoiled sites, and from 0.0 to 285.0 ng/g wet weight (ppb) at oiled sites. Oil concentration in mussels followed a similar pattern, ranging from 19.8 to 37.8 ng/g (ppb) at unoiled sites, and from 24.7 to 4,366.0 ng/g (ppb) at oiled sites (Table 20; Figure 37). Estimates of total aromatic hydrocarbon concentrations followed the same trend as estimated total oil concentration.

Mean levels of aromatic hydrocarbons and phytane in mussels from oiled sites were similar to those found in crude oil from the *T/V Exxon Valdez*: both had aromatics dominated by phenanthrene with lesser levels of dibenzothiophenes, fluorenes, and phytane (Figures 4 and 36). None of the oiled or unoiled samples appeared to have been contaminated by refined hydrocarbons, which contain napthalenes as the dominant aromatic component.

By 1990, mean levels of aromatic hydrocarbons and phytane in mussels from all but one (Smith Island) oiled sites, as well as all unoiled sites, were much reduced (Table 16; Figures 38 and 39). Total aromatic hydrocarbons ranged from 22.1 to 23.3 ng/g wet weight (ppb) at unoiled sites, and from 29.5 to 623.3 ng/g (ppb) at oiled sites. No phytane was detected in mussels from all three unoiled sites and two of the five oiled sites. In samples from the three oiled sites where it was detected, phytane ranged from 9.0 to 98.8 ng/g (ppb). Oil concentration ranged from 8.4 to 10.7 ng/g (ppb) at unoiled sites, and from 11.2 to 1,439.8 ng/g (ppb) at oiled sites. The composition of the elevated levels of aromatic hydrocarbons found in mussels collected at Smith Island was again similar to that of *T/V Exxon Valdez* crude oil (Figures 4 and 38).

DISCUSSION

Exposure of Herring to Oil

Adult Herring

In 1989, adult herring in PWS probably migrated from wintering to spawning areas in a path counter to the trajectory of the oil spilled from the *T/V Exxon Valdez* (Figure 9). Although some fishes actively avoid oil and much of the evidence measuring avoidance in the natural is circumstantial (Rice 1985). If water concentrations of pollutants are below acutely toxic and fish are highly motivated to migrate as are herring in spawning condition, they may not avoid the contaminated areas. In the case of the herring larvae, they most certainly cannot avoid oil (Rice 1985). While we were unable to collect herring until two weeks after the *Exxon Valdez* oil spill had occurred, we expected that samples would have high levels of unmetabolized aromatic hydrocarbons since herring have been shown to metabolize oil poorly and accumulate hydrocarbons rapidly, especially in immature ovaries (Rice et al. 1987a, 1987b; Sid Korn, NMFS, personal communication). However, not only were PAH levels in gut (147.7 ppb) and gonad (73.4 ppb) tissues of herring collected in oiled areas not much greater than levels found in herring collected in unoiled areas (138.0 ppb, gut; 48.7 ppb, gonad), but these levels were also near the lower limits of procedures used for detection (Jeff Short, NMFS, personal communication).

Results of histopathology and parasitology studies of herring collected in the spring of 1989 provide stronger evidence that adult herring were exposed to *Exxon Valdez* oil. Severe hepatic necrosis was identified in 20% of herring examined from oiled sites (Naked Island and Rocky Bay), but in none of the herring examined from unoiled sites (Fairmont and Galena Bays; Marty et al. 1993). Hepatic necrosis is indicative of recent exposure to a toxin and was found in fish 9 months after the *Amoco Cadiz* spill in 1978 (Haensly et al. 1982). Recent oil exposure experiments with adult herring (Sid Korn, NMFS, personal communication) produced hepatic necrosis identical to that found in PWS herring in 1989 (Marty et al. 1993). Moles et al. (1993) reported that nematode abundance in the guts of herring collected in oiled areas of PWS in 1989 was much lower than that found in the guts of herring from unoiled areas. These investigators found that the missing nematodes had migrated from the gut cavity into the surrounding musculature, and they were able to duplicate this phenomenon in the laboratory by exposing herring to known levels of *Exxon Valdez* crude oil. Levels of gut parasites in PWS herring from oiled areas had returned to a more normal baseline level in samples collected in 1991. Unfortunately, while information from histopathology and parasitology studies demonstrate that PWS herring were exposed to and injured by *Exxon Valdez* oil, tissue samples were not collected in a manner that allowed rates of exposure or damage to be expanded to the population level.

Further evidence that adult herring were exposed to oil was obtained from bile samples collected incidentally to sampled obtained from cod and halibut during the summer of 1989, bile samples collected from spawning herring in the spring of 1990, and by histopathology samples from herring caught in the fall of 1990. Herring sampled in the summer of 1989 came from a heavily oiled bay on Knight Island, a feeding area. Herring sampled in the spring of 1990 came from Outside Bay (an oiled area on Naked Island), Port Chalmers (an area adjacent to the path of the oil and near other heavily oiled areas on Montague Island), and Wells Bay (near the unoiled North Shore area). Herring samples

taken both years, from unoiled as well as oiled areas, had whole fraction oil in their bile (Carol Ann Manen, NMFS, personal communication). This suggested that even herring spawning in unoiled locations could have been exposed to oil later in the year during their migration. For example, large aggregations of herring occur in the Green Island trench between Montague and Green Islands during fall and winter, where they are the target of a food and bait fishery. This area was within the path of the oil spill, and herring sampled from this area during the 1990 fall fishery had a higher incidence of hepatic macrophage aggregates (Marty et al. 1993) and greater MFO and P450 induction Stegemen (1992), both indicators of oil exposure, than herring sampled from Knowles Head, an unoiled area. These differences were greatly reduced, or not found, by the spring of 1991.

Routes of oil exposure in adult herring in 1989 and 1990 include direct exposure to floating and suspended oil as well as consumption of oiled prey. In the adult dose-response study, numbers of lesions were elevated in herring directly exposed to WSF and fed oiled pellets compared to controls. These lesions were similar to ones observed in wild-caught fish in 1989 and 1990.

Juvenile Herring

Juveniles as well as adult herring could also have been exposed to oil during 1989 and 1990. Based on the spatial and temporal distribution of historical catches in reduction, food, and bait fisheries (Reid 1971; Rounsefell and Dahlgren 1932), juvenile as well as adult herring appear to spend the summer feeding and rearing in plankton-rich, nearshore bays and passes in PWS. These areas were overlapped by the trajectory of oil to a greater extent than were spawning areas: approximately 80% of historic summer fishing areas were oiled. Herring in the 1988 year class, which were juveniles during the *Exxon Valdez* spill, comprised 64.6% of the spawning population, by number, in 1992. This year class was forecasted to comprise 84.0% of the 1993 run biomass (Fritz Funk, Alaska Department of Fish and Game, personal communication). Evidence of long term reproductive effects of oil exposure on the 1988 year class is covered by Kocan et al. (1993) in the 1992 pilot study on reproductive impairment. The 1988 year class were collected as returning 4-year-old spawners.

Herring Eggs and Larvae

Injuries from oiling of herring eggs and larvae in 1989 were probably not due to the WSF since levels of these toxicants were likely below the effects concentrations (EC; as low as 10 ppb in Mironov 1969) known to affect these life stages. Although Neff (1991) and Short and Rounds (*In prep*) recorded concentrations close to or exceeding 10 ppb, average concentrations were much lower. Most experiments on effects of oil on fish have been conducted with a WSF solution produced by shaking oil in water and then discarding the surface film prior to exposure. Few researchers have addressed the toxicity of the surface film or "microlayer" of an oil spill to which organisms in tidal and shallow nearshore areas could be exposed. Kocan et al. (1987) and Cross et al. (1987) found that surface films were 10 to 10,000 times more toxic than the WSF a few inches below the surface and could induce reduced hatching success and increased larval abnormalities in fishes. Pearson et al. (1985) stated that oil dispersed by turbulence produces suspended oil droplets that can adhere to intertidal and subtidal herring eggs through wave action. They also reported that unfiltered, undispersed oil droplets produced the most severe impacts on herring eggs in experiments comparing the effects of filtered and unfiltered, chemically-dispersed and undispersed crude oil. The

amount of oil adhering to eggs was the primary factor determining the frequency of abnormal larvae in these experiments. Short and Heintz (*in prep*) believe, based on data on oiled mussels, that suspended oil droplets and oiled particles were prevalent throughout the oil trajectory both inside and outside of PWS in 1989.

Given the wave action, currents, and 8 m tidal range in PWS and due to their nearshore distribution, it is highly likely that PWS herring embryos and larvae were directly oiled in 1989 on or near the spawning grounds both by microlayer oil and suspended oil droplets. Of 1,349 herring larvae captured during a June 1989 larval fish survey, 87% were collected from oiled areas around Knight and Montague Islands (Norcross and Frandsen 1993). Furthermore, 50% of all herring larvae captured were collected due south of a heavily oiled beach on Green Island that leached oil for nearly a year. Herring and other forage fishes such as juvenile pink salmon (Mark Willette, Alaska Department of Fish and Game, personal communication) commonly appear to concentrate in the southwestern portion of PWS in the summer, perhaps due to flow patterns which concentrate food in this area. It is unlikely that consumption of oiled prey would have affected large numbers of larvae since zooplankters can rapidly depurate oil (Carls 1987) and larvae fed contaminated prey in laboratory experiments showed less oiling effects than those directly exposed to WSF (Rice et al. 1987b).

Eggs within the ovaries of adults may also have been exposed to oil if adults traversed oil slicks. Oil is absorbed into eggs through the lipid-rich yolk (Struhsaker 1977) and the PAH fraction is highly lipophilic (Connell and Miller 1984; Rice et al. 1987a). While herring sampled two weeks after the spill showed very low levels of oil, no samples were collected just prior to spawning. Separating injuries to eggs and larvae due to direct exposure at spawning sites from those due to indirect exposure of their parents will be difficult.

The Prince William Sound Herring Population

Run Timing and Spawn Distribution

Adult herring have been made to spawn prematurely as a result of exposure to benzene, and ovarian eggs have been shown to accumulate benzene levels twice that of newly spawned eggs (Struhsaker 1977). Spawning activity in PWS did occur earlier than usual following the 1989 oil spill, and fewer eggs were laid than expected. While it is possible this occurred as a result of exposure of ovarian eggs to oil, egg retention was not measured in 1989 and a single sample of females collected prior to spawning, and two weeks after the oil spill, from oiled and unoiled areas did not show any evidence of egg resorption or attrition (Hose 1993).

Annual herring run timing in PWS varies, but the mean date of spawning for most years generally falls within five days of the overall mean date (Figure 11). The mean spawning date for 1989 was the earliest ever recorded, being seven days earlier than the overall mean date, but factors other than oil exposure, such as water temperature (Ware and Tanasichuk 1989), can also affect spawning behavior. Water temperature in Gulf of Alaska waters adjacent to PWS was unseasonably warm in April 1989, while winter temperatures had been colder than average (Mark Willette, Alaska Department of Fish and Game, personal communication). Because the relationship between winter (positively correlated) and spring (negatively correlated) Gulf of Alaska sea-surface temperature anomalies and the mean date of spawning in PWS was found to be signigicant, we feel that environment factors affected spawning timings in

1989. We cannot distinquish a potential oil effect from the environmental effect.

The distribution of spawn among the five major spawning areas also changed slightly from year to year (Figures 1 and 12). There was a general shift of spawning from the Naked Island and North Shore areas to the Montague Island and Northeast areas over the period of the study. Along with these changes we also noted a decline in total shoreline miles of spawn since 1988, which was accompanied by an increase in both spawn width and egg densities from 1988 to 1991. This trend may have reversed in 1992 when an increase in shoreline miles of spawn and a decline in spawn width and egg density was noted. Again, we have no evidence to suggest that oil influenced these spawning area shifts.

Spawning Behavior and Success

The spawn deposition escapement biomass estimates are in general agreement with the ASA estimates of biomass except for 1989 (Figure 17). Whether oil caused egg retention and lower than expected egg deposition as a result is not known. In Puget Sound, ovarian maturation was inhibited in English sole from areas with high sediment aromatic hydrocarbon (Johnson et al. 1988) and fish from these areas that did mature often failed to spawn after hormonal treatment to induce spawning (Casillas et al. 1991). Reproductive impairment, including reduced estradiol or sex steroid levels, was found in the sole that showed evidence of aromatic hydrocarbon exposure. In order to measure egg retention, samples should be stratified temporally and spatially. In addition, fish should be sampled after all spawning events have ceased. This was not done in 1989. There is no evidence of egg resorption or atretic ova in the single sample of adult fish taken two weeks after the spill in 1989 (Hose 1993). Measuring oocyte-loss and egg retention was added to the 1990 study, but was not an objective of the original 1989 study plan plan. In addition, no samples of plasma in adult herring were collected to measure potential reductions in steroids that could have affected herring spawning. Therefore, the cause of the large disparity between the spawn deposition survey and ASA estimate of total eggs will never be known. However, it seems unlikely that the timing of the spill and this disparity could occur on the same year.

Population Effects of Oil Spill Injuries

Some evidence exists to suggest that 1989 year class of herring was seriously impacted by the oil spill. Fritz Funk (Alaska Department of Fish and Game, Juneau, personal communications) found a strong correlation between the number of 3-year-old recruits in PWS and Sitka Sound, Southeast Alaska. Recruitment in 1989, however, was very different between the two areas. Age 3 herring (1989 year class) abundance in PWS in 1992 was one of the lowest ever recorded while abundance of this age class in Sitka Sound was about average. With warm ocean temperatures and a high zooplankton index for PWS in 1989 (McGurk 1990a), there was no reason to expect low larval survival all other factors being equal. Oil exposure probably caused great mortality of eggs and larvae produced in 1989 (Hose 1993). While there is great variability in the relationship between number of eggs deposited and resulting number of recruits to the spawning population three and four years later, Lasker (1985) and Smith (1985) found good relationships between numbers of larvae surviving and the number of recruits for various clupiods.

Sublethal effects of the oil exposure in 1989 and 1990 on adult and juvenile herring hatched prior to 1989 have not been thoroughly examined. A pilot study conducted in 1992 (Kocan and Mehl 1993) suggested that herring which

spawned in previously oiled areas may be reproductively impaired. Unfortunately, this study was not continued in 1993. The 1992 year class will not begin recruiting to the spawning population until 1996, but without a properly designed monitoring program and results from a 1993 reproductive impairment study, it will be difficult to link future changes in recruiting patterns to long-term injury in adults from exposure to oil.

Trends in Fecundity, Egg Size, and Gonad Weight

We felt that effects of the oil spill on fecundity, egg size and gonad weight would have been manifested most strongly one or two years after the spill (1990 and 1991) and then would have diminished over the preceding years. While some evidence for oiling effects was obtained, it was difficult to separate this from other causes. Fecundity has been found to be proportional to body weight for Atlantic herring stocks in the Baltic Sea (Oyaveyer 1983), Pacific herring stocks in British Columbia (Tanasichuk and Ware 1987; Ware 1985; Hourston *et al.* 1981; Kristofferson and Gillman 1986) and for Pacific herring sampled from the Green Island area in PWS (Paulson and Smith 1977). We found that fecundity, egg weight, and gonad weight were all linearly related to herring body weight. Therefore, we included body weight as a covariate in ANOVA and ANCOVA models for fecundity, egg weight, and gonad weight to see if it would explain (i.e. remove) significant year-area interactions. While inclusion of body weight did not remove the interaction, it did diminished its magnitude. The addition of length and age as covariates also did not explain year-area interactions. While some combination of these factors might explain area and year effects, we did not explore this possibility. Stock specific differences may also account for differences in fecundity, egg weight, and gonad weight (Katz 1947; Zijlstra 1973), but we had no way to determine this for PWS herring.

Environmental factors have also been shown to influence fecundity (Tanasichuk and Ware 1987). We found that fecundity, egg weight, and gonad weight were highly correlated with SSTA data prior to spawning (Figure 27), but could not use SSTA data as a covariate in an ANOVA model since they were completely confounded with year. Tanasichuk and Ware (1987) found that mean sea surface temperatures about two or three months prior spawning best accounted for variations in size-specific fecundity for Pacific herring in British Columbia. We found that egg weight was correlated to SSTA's 4 to 9 months prior to spawning but that fecundity was correlated to SSTA's 13 to 15 months prior to spawning (Figure 12). Based upon these relationships, we suggest fecundity of herring in PWS is primarily determined by body weight and is most strongly affected by sea surface temperatures 13 to 15 months prior to spawning, while egg weight is primarily determined by fecundity and body weight and is most strongly affected by sea surface temperatures 4-9 months prior to spawning.

We also found that fecundity decreased as water temperatures increased. This is opposite to reported effects of water temperature upon Atlantic herring (Baxter 1959, Cushing 1967, Messieh 1976) and Pacific herring in the southern portion of their range (Paulson and Smith 1977), but is in agreement with the finding for Pacific herring in British Columbia (Tanasichuk and Ware 1987).

Egg Loss Study

Based on results of our study, we feel that the best estimate of daily egg loss for PWS herring is 2.1%, the mean estimate for all areas and depths for both years, excluding data from 1.65 m. Haegele and Schweigert (1991) estimated total egg loss from deposition to hatching to be 58% in Lambert Channel in the Strait of Georgia, British Columbia, with greater egg loss in

shallow (92%) than in deep (46%) water. These total egg loss estimates were very similar to our estimates of total egg loss: 91.2% in shallow water and 50.4% in deep water.

Accurate estimates of egg loss are needed to provide accurate estimates of spawning biomass from spawn surveys. For example, if actual egg loss was 50% and it was estimated to be 10%, the resulting spawning biomass estimate would have to be increased by 80.2% to equal the actual spawning biomass. The average number of days between spawning and spawn surveys in PWS was between 5 and 7 days in 1990 and 1991. Based upon this, total egg loss prior to spawn surveys would have been between 10.5% and 14.7%, which is very similar to our assumed estimate of 10% used to calculate biomass estimates for this report. In future years, we recommend multiplying the actual number of days between spawning and spawn surveys by 2.1% to estimate total egg loss, unless a large proportion of eggs are deposited in the upper intertidal area. In this situation, a daily egg loss estimate of 3.8% should be used to estimate total egg loss.

Although the main cause of egg loss reported in previous studies has been predation by sea birds, invertebrates, marine mammals, and other fish (Outram 1958;1 Steinfeld 1971; Haegele and Schweigert 1989 and 1991). Estimates of total egg loss due to predation have ranged from 7% or 8% for subtidal Atlantic herring spawn (Tibbo *et al.* 1963; Caddy and Iles 1973) to 70% for upper intertidal zone Pacific herring spawn (Steinfeld 1971). Haegele and Schweigert (1989 and 1991) estimated that birds, invertebrates, and whales in Barkley Sound and Lambert Channel, British Columbia consumed 7.1% to 20% of available Pacific herring spawn. Estimates of egg loss due to predation in most studies probably represented less than half of the total loss, suggesting that either predation was underestimated or that losses due to other factors had been overlooked. We think that the primary cause of egg loss in PWS is wave and tidal action. This would account for the differences observed between our shallowest depth (1.65 m) and the rest of the sampled depths. While we have observed predation of eggs by sea birds and invertebrates, it does not appear to be great.

Some of our data suggest that eggs were transported from intertidal to subtidal areas by wave or tidal action, since we recorded large egg losses at the 1.65 m depth at the same we recorded increases in egg densities at lower depth zones. We could find no information in the literature concerning the fate of eggs transported into subtidal areas, although it is more likely that unattached egg masses would eventually be washed up on shore or carried out to sea. Hart and Tester (1934) noted that about 40% of the spawn at Departure Bay, British Columbia, washed ashore after a storm and about 70% of these eggs were dead. Steinfeld (1971) estimated that spawn in upper intertidal areas of Yaquina Bay, Oregon, suffered high mortality due to a high frequency of unfertilized eggs and desiccation due to exposure to air.

Since our estimates of daily loss of eggs were highly variable both within and among areas, it is possible that gains in eggs noted along some transects were actually sampling artifacts. Johannessen (1986) and Haegele and Schweigert (1989, 1991) all reported that variances of egg densities in their studies were so great that it was difficult or impossible to obtain significant regressions of egg density on egg age. The uneven distribution of spawn makes it difficult to accurately estimate changes in egg density. Reduction in the variance of egg density estimates might be obtained with the use of larger quadrats, an increase in the number of quadrats sampled, or other improvements in experimental design.

We did not try to determine whether the oil spill affected egg loss, although we found that egg loss rates were generally greater in the oiled areas. There is no evidence that oil affects the adhesive qualities of the eggs and could cause increased egg loss. In addition, our work was conducted in 1990 and

1991 when the quantity of oil in the water column was much reduced. Therefore, we feel that differences in egg loss among sites were probably due to differences in the physical characteristics of each area rather than oil spill effects. The oiled areas in PWS were generally more exposed and unprotected than unoiled areas, which would be expected since the oil was carried on water currents and contaminated the most exposed and unprotected areas. Since the oil spill killed a large number of sea birds, egg loss due to predation may actually have been lower in 1989 and subsequent years.

Hydrocarbon Sampling and Analysis

Reported concentrations of oil contaminants in the water column following oil spills depend upon the amount and type of oil spilled, weather and water conditions at the time of the spill, where samples were collected, and how measurements were made. Pearson et al. (1985) stated that the maximum expected hydrocarbon concentration in the water column following an oil spill should generally range between 0.2 and 0.65 ppm, but could exceed 1.0 ppm if turbulent conditions physically dispersed oil into the water column. Føyn and Serigstad (1989) felt realistic upper level concentrations of oil likely to be present after a spill ranged from 90 ppb to 0.3 ppm PAH, while Connell and Miller (1984) suggested an upper limit of 0.1 ppm of soluble aromatic compounds for polluted estuaries and coastal areas. Measurements of hydrocarbon concentrations after actual spills have generally been within these expected ranges. Hydrocarbon concentrations exceeding 1 ppm were recorded in water entering an estuary after the *Amoco Cadiz* spill, while concentrations of 0.3 ppm were measured in the upper 5 m of the water in another North Sea spill (Carls 1987). Hydrocarbon concentrations ranging from 0.2 to 0.3 ppm were detected in seawater a month after a 2,000 tons spill of Iranian crude oil that produced concentrations ranging from 0.002 to 0.8 ppm (0.1 ppm average) just after the spill (Grahl-Nielson et al. 1978).

Dissolved PAH's are acutely toxic at levels ranging from 0.1 to 0.5 ppm, but concentrations after a spill are generally below this ranges which would suggest that sublethal effects on organisms would be a more likely result (Connell and Miller 1984). Hydrocarbons only affect organisms when these compounds persist at the concentration that measurable effects (EC) or long enough for measurable bioaccumulation to occur. Behavioral effects, such as feeding, mating, and physiological function, occur in adults at concentrations of soluble aromatics ranging from 10 ppb to 1.0 ppm, growth and reproductive effects generally start to occur at 100 ppb, while lethal effects for larvae and juveniles begin to occur at 1.0 ppm and for adults at 100.0 ppm (Connell and Miller 1984).

Two weeks after the *Exxon Valdez* spill, Short and Rounds (*In prep*) measured dissolved PAH concentrations of 6 ppb between northern Montague, Smith, and Knight Islands in PWS. In addition, they found that oiled beaches acted as reservoirs which continued to leach PAH's into PWS as long as five weeks after the spill. Furthermore, due to the sample extraction method used, these investigators felt that PAH concentration was actually greater than measurements they obtained, and that total aromatic hydrocarbon concentrations probably exceeded 10 ppb (Alaska's clean water limit for PAH) after the spill. This concentration is at the lower end of the EC for embryos and larvae of many marine fishes (Jeep Rice, NMFS, personal communication). Effects of long term exposure to these concentrations in the water column is largely unknown.

Measuring the toxicity of a spill using estimates of dissolved oil concentrations may be a misleading indicator of potential injuries to organisms, since suspended, non-dissolved oil droplets created by wave turbulence can heavily impact shallow water communities (Pearson et al. 1985),

and organisms near an oil spill or under a surface sheen are exposed to continuously changing concentrations of petroleum hydrocarbons (Neff 1990). Effects of an oil spill can also be exacerbated by environmental conditions such as low annual mean water temperatures, low oxygen content in bottom waters, and low salinities which aggravate pollution problems and prolong renewal times (Lindén 1978). Arctic organisms may be even more vulnerable to oil spills than temperate or tropical organisms since aromatic hydrocarbons persist longer in cold water (Rice 1985).

The level of histopathological injury in adult herring captured in PWS after the 1989 spill closely matched injury levels found in adult herring exposed to between 0.68 and 1.20 ppm total aromatics (using *Exxon Valdez* crude) in paired adult dose-response experiment conducted in 1991 at Auke Bay (Marty et al. 1993). In these same dose-response experiments, adult herring had to be exposed to 1.2 ppm total aromatics for 12 days to create the same absence of parasites noted in the gut of adult herring in oiled areas of PWS in 1989 (Moles et al. 1993). Therefore, it appears that some portion of the PWS herring population was exposed to very high concentrations of PAH.

Sublethal effects on growth and reproduction are the most likely effects of a spill on aquatic organisms and are probably most severe for estuarine or shallow-water populations (Rice 1985). Larval and juvenile organisms, due to their size, limited ability to avoid spill contaminants, and fast growth rates, might be most prone to these effects, and would be expected to suffer deformities and mortalities (Connell and Miller 1984). Since predicted concentrations and toxicities of the *Exxon Valdez* oil spill fell within levels that were known to cause sublethal injuries to larval herring, herring embryos and larvae in the spill path would likely have been injured. Furthermore, adult and juvenile herring in PWS were probably exposed to oil longer than time periods used in existing laboratory studies, so long term sublethal effects are difficult to predict.

Since it is difficult to measure hydrocarbon concentrations in the water column and these measurements may be poor indicators of injury to organisms, we chose to use mussel tissue samples to index oiling exposure. *Exxon Valdez* crude oil was found in mussels down to a depth of 40 m downstream from Bligh Reef in 1990 (Short and Rounds *in prep*). Mussels must have been contaminated by filtering oil droplets or oil adhering to particulate matter, water chemistry tests did not indicate the presence of oil in the water column. Other investigators also found indications that oil was present well below the water (Dave Sales, DEC, personal communication; Howard Feder, University of Alaska, personal communication; Doug Wolfe, NOAA, personal communication). This suggests that other invertebrates and fishes might also have been exposed to or ingested oil below the surface in 1989 and 1990. A high correlation of indices of injury in larval herring and level of oil in mussels sampled from the same sites was found by Hose (1993), while no correlation between the same indices of injury and water column chemistry results was found by Pearson et al. (1993).

Finally, pollutants present in an area prior to an oil spill can have synergistic and antagonistic interactions with toxic substances from the spill (Karinen 1988; Rice et al. 1987). PWS, however, was generally considered to be relatively pristine prior to the *Exxon Valdez* spill (Babcock and Karinen 1991; Karinen et al. 1993). Although low levels of contaminants from small fuel spills, ballast water discharges, and fuel-combustion discharges were identified in Rocky Bay (Karinen et al. 1993), mussel tissue collected from this area in 1989 were found to contain concentrations of PAH (from 0.12 to 2.28 ppm total PAH wet wt) more than ten times greater than those measured from mussel tissue collected from this area prior to the spill. Two of our three study transects in Rocky Bay in 1989 showed high levels of aromatic hydrocarbons which were indicative of *Exxon Valdez* crude oil (Figure 36). Hydrocarbon analyses can readily distinguish crude oil from low level chronic

fuel sources (Jeff Short, NOAA-NMFS, Auke Bay Lab, Juneau, personal communication). So there is little doubt that oil measured in mussels collected at herring study sites in Rocky Bay was from the *Exxon Valdez* spill.

CONCLUSIONS

1. The oil spill occurred days before herring spawning migration commenced, but peak concentrations of oil in the water coincided with the peak date of spawning in mid-April. Dissolved oil in the waters of PWS reached concentrations exceeding the 10 ppb Alaska clean water limit, but average concentrations remained low. Suspended oil-water droplets and oiled particulate matter were prevalent throughout the oil trajectory zone in PWS in 1989 and to a lesser extent in 1990 and were biologically available to intertidal, subtidal, and pelagic marine organisms.

   Recommendation: It has taken four years to resolve the fate, toxicity, and concentrations of oil in PWS and there are still disagreements in specific areas. Analysis and synthesis of oil spill impact study results should continue allowing for the incorporation of this new information. These synthesis products should be integration between studies and species and incorporated in a PWS ecosystem modeling exercise. The ecosystem model would be useful both in monitoring and response planning.

2. Adult herring migrating to the spawning grounds in 1989 were exposed to oil, although it is not known what proportion of the population was exposed and the dose of exposure. Exposure to oil continued throughout 1989 and into 1990. Internal tissues were damaged but the short and long term effects of the oil exposure are speculative. There may have been a short term effect which inhibited egg deposition and a long term effect that caused reproductive impairment (reduced survival of offspring) but conclusive evidence is missing. Studies conducted from 1989 to 1992, including egg loss and fecundity, did improve the accuracy of the biomass estimates and resulting forecasts, but were dropped in 1993.

   Recommendation: Continue to monitor the PWS herring population with accurate assessment tools, particularly the spawn deposition survey.

   Continue to study reproductive impairment in wild-caught adult herring exposed to oil in 1989 and 1990 through a laboratory study in which success of the survival of the offspring are measured.

   Implement a herring tagging study that would resolve gaps in understanding stock discreetness and distribution and would improve the interpretation of adult injury information.

3. Eggs were deposited in oiled areas in 1989. Larvae hatched from exposed embryos and suffered reduced survival as a result. The exact dose of oil that embryos received is unknown, but levels of oil measured in mussels collected adjacent to the eggs were significantly correlated to rates of sublethal injury in the larvae. Surviving larvae were concentrated in oiled areas by the same currents that drove oil out of

the Sound. This 1989 year class returned at lower levels than expected in 1992 and reduced larval survival due to exposure to oil was the most likely cause cited.

Larvae hatching from eggs deposited in oiled areas in 1990 continued to show increased sublethal injury. However, it is not known how that may have affected the survival of the 1990 year class.

Recommendation: Conduct surveys on larval and juvenile herring that could resolve gaps in understanding of survival, growth, and distribution. Coordinate and integrate such surveys with other forage fish studies.

Initiate a hydroacoustic survey allowing the PWS population to be tracked and sampled. This survey could also be coordinated with the study of other forage fish.

4. The effects of environmental parameters, such as sea-surface temperatures in the Gulf of Alaska, on PWS herring run timing, growth, fecundity, and recruitment have only recently been examined. In addition, the relative influences of environmental and density-dependent factors on stock abundance and age composition are unknown. In order to speculate on the effects of oil exposure on the population of herring in PWS, we must invest some effort in understanding these phenomena more accurately.

Recommendation: Continue the analysis of the 20 year time series on PWS herring abundance, size, and age composition data and develop a model with environmental data that can explain much of the variability. Incorporate this with the ecosystem approach monitoring plan.

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Table 1. Herring biomass estimates in tonnes from aerial survey by area and date, Prince William Sound, 1989

	Southeastern				Northeastern					North Shore			Naked Island					
Date	Simpson Sheep & Islands	Hinchin-Brook Island	Port Gravina	Area Total	Port Fidalgo	Tatitlek Area	Bligh Island	Valdez Arm & Port	Area Total	Freemantle Granite Pt.	Granite Pt. Esther Pass.	Area Total	Naked Island	Knight Island	Area Total	Montague Island	PWS Total	Date
03/30	0.0		0.0	0.0	0.0	136.1		0.0	136.1			0.0	0.0		0.0		136.1	03/30
03/31	0.0		0.0	0.0	9.1	181.4	0.0	0.0	190.5	0.0	0.0	0.0	0.0	0.0	0.0	0.0	190.5	03/31
04/01	0.0			0.0	90.7	344.7		0.0	435.4	0.0	952.5	952.5			0.0		1,387.9	04/01
04/02	0.0		0.0	0.0	0.0	54.4		18.1	72.5	0.0	1,324.5	1,324.5	226.8	0.0	226.8	0.0	1,623.8	04/02
04/03	0.0		0.0	0.0	9.1	335.7		49.9	394.7	0.0	1,183.9	1,183.9	145.1	0.0	145.1		1,723.7	04/03
04/04	0.0		0.0	0.0	362.9	0.0		0.0	362.9	0.0	1,868.8	1,868.8	117.9	72.6	190.5	0.0	2,422.2	04/04
04/05	0.0		63.5	63.5	1,632.9	526.2		0.0	2,159.1	27.2	4,227.5	4,254.7	499.0	0.0	499.0	0.0	6,976.3	04/05
04/06	0.0		0.0	0.0	272.2	898.1	0.0	281.2	1,451.5	290.3	7,139.6	7,429.9	961.6	0.0	961.6	0.0	9,843.0	04/06
04/07	9.1	0.0	0.0	9.1	753.0	571.5		412.8	1,737.3	3,728.5	8,781.6	12,510.1	1,369.9		1,369.9		15,626.4	04/07
04/08	0.0		0.0	0.0	81.6	435.4	18.1	4,998.6	5,533.7	1,732.7	6,749.5	8,482.2	1,905.1	81.6	1,986.7	0.0	16,002.6	04/08
04/09	0.0	0.0		0.0	176.9	281.2		684.9	1,143.0	1,102.2	7,538.7	8,640.9	1,474.2		1,474.2	0.0	11,258.1	04/09
04/10	No Survey Today																	04/10
04/11					970.7	308.4	235.9	2,630.8	4,145.8	99.8	1,778.1	1,877.9	2,785.1	0.0	2,785.1	0.0	8,808.8	04/11
04/12	0.0	0.0		0.0	1,723.7	371.9	0.0	1,850.7	3,946.3	843.7	1,877.9	2,721.6	4,340.9		4,340.9	24,947.6	35,956.4	04/12
04/13					208.7	90.7	163.3	353.8	816.5	489.9	263.1	753.0	880.0	72.6	952.6	18,651.7	21,173.8	04/13
04/14	0.0			0.0	163.3	9.1	113.4	267.6	553.4	140.6	95.3	235.9	571.5	72.6	644.1	13,430.9	14,864.3	04/14
04/15			18.1	18.1	36.3	0.0	0.0	9.1	45.4	0.0	18.1	18.1	462.7	190.5	653.2	12,183.5	12,918.3	04/15
04/16	No Survey Today																	04/16
04/17	0.0	0.0	18.1	18.1	13.6	0.0	0.0	29.9	43.5	0.0	27.2	27.2	0.0	136.1	136.1	1,696.4	1,921.3	04/17
04/18		63.5	0.0	63.5	0.0	0.0	0.0	9.1	9.1	0.0	0.0	0.0	190.5	154.2	344.7	4,799.0	5,216.3	04/18
04/19	0.0			0.0	0.0				0.0	0.0	4.5	4.5	0.0	145.1	145.1	880.0	1,029.6	04/19
04/20			0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	127.0	127.0	0.0	127.0	04/20
04/21									0.0	0.0		0.0	0.0	117.9	117.9	0.0	117.9	04/21
04/22				0.0			0.0		0.0			0.0	0.0	54.4	54.4	0.0	54.4	04/22
04/23	No Survey Today																0.0	04/23
04/24	No Survey Today																0.0	04/24
04/25	No Survey Today																0.0	04/25
04/26	No Survey Today																0.0	04/26
04/27	No Survey Today																0.0	04/27
Total	**9.1**	**63.5**	**99.7**	**172.3**	**6,504.7**	**4,544.8**	**530.7**	**11,596.5**	**23,176.7**	**8,454.9**	**43,830.8**	**52,285.7**	**15,930.3**	**1,224.6**	**17,154.9**	**76,589.1**	**169,378.7**	**Total**
Peak Aerial Estimate:	**9.1**	**63.5**	**63.5**	**136.1**	**1,723.7**	**898.1**	**235.9**	**4,998.6**	**7,856.3**	**3,728.5**	**8,781.6**	**12,510.1**	**4,340.9**	**190.5**	**4,531.4**	**24,947.6**	**Total Peak Aerial Estimate 49,981.4**	

Table 2. Herring mile days of milt sightings from aerial surveys by area and date, Prince William Sound, 1989

	Southeastern				Northeastern					North Shore			Naked Island					
Date	Simpson Sheep & Islands	Hinchin-Brook Island	Port Gravina	Area Total	Port Fidalgo	Tatitlek Area	Bligh Island	Valdez Arm & Port	Area Total	Freemantle Granite Pt.	Granite Pt. Esther Pass.	Area Total	Naked Island	Knight Island	Area Total	Montague Island	PWS Total	Date
03/30	0.0		0.0	0.0	0.0	0.0		0.0	0.0			0.0	0.0		0.0		0.0	03/30
03/31	0.0		0.0	0.0	0.0	0.2	0.0	0.0	0.2	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.2	03/31
04/01	0.0			0.0	0.0	0.0		0.0	0.0	0.0	0.0	0.0			0.0		0.0	04/01
04/02	0.8		0.3	1.1	0.0	0.0		0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	1.1	04/02
04/03	0.0		0.0	0.0	0.0	0.2		0.0	0.2	0.0	0.4	0.4	0.0	0.0	0.0		0.6	04/03
04/04	0.0		0.0	0.0	0.0	0.0		0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	04/04
04/05	0.0		0.0	0.0	0.0	0.0		0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	04/05
04/06	0.0		0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	04/06
04/07	0.0	0.0	0.0	0.0	0.0	1.4		0.0	1.4	0.0	1.2	1.2	0.0		0.0		2.6	04/07
04/08	0.0		0.0	0.0	0.0	1.5	0.0	0.0	1.5	0.0	0.0	0.0	0.0	0.0	0.0	0.0	1.5	04/08
04/09	1.0	0.0		1.0	0.4	6.0		0.0	6.4	0.0	3.2	3.2	1.3		1.3	0.0	11.9	04/09
04/10	No Survey Today																	04/10
04/11					0.0	5.0	2.0	0.0	7.0	0.0	8.0	8.0	5.0	0.0	5.0	0.0	20.0	04/11
04/12	0.0	0.0		0.0	0.1	1.0	5.2	0.0	6.3	0.0	10.0	10.0	6.0		6.0	0.2	22.5	04/12
04/13				0.0	0.0	15.3	4.3	0.3	19.9	0.0	13.0	13.0	6.0	0.0	6.0	2.2	41.1	04/13
04/14	0.0			0.0	0.0	1.3	0.5	0.1	1.9	1.3	6.6	7.9	0.9	0.0	0.9	2.6	13.3	04/14
04/15			0.0	0.0	1.1	2.0	1.0	0.8	4.9	2.1	7.7	9.8	0.0	0.0	0.0	2.8	17.5	04/15
04/16	No Survey Today																	04/16
04/17		0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.3	0.1	0.4	17.1	17.5	04/17
04/18		0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.1	0.1	0.2	1.5	0.0	1.5	19.0	20.7	04/18
04/19	0.0			0.0					0.0	0.4	0.2	0.6	0.0	0.0	0.0	6.2	6.8	04/19
04/20					0.0	0.0	0.0	0.0	0.0	0.5	0.0	0.5	0.0	0.0	0.0	7.0	7.5	04/20
04/21										0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	04/21
04/22		0.0	0.0	0.0									0.0	0.0	0.0	0.0	0.0	04/22
04/23	No Survey Today														0.0			04/23
04/24	No Survey Today														0.0			04/24
04/25	No Survey Today														0.0			04/25
04/26	No Survey Today														0.0			04/26
04/27													0.0	0.0	0.0	0.0	0.0	04/27
Total	**1.8**	**0.0**	**0.3**	**2.1**	**1.6**	**33.9**	**13.0**	**1.2**	**49.7**	**4.4**	**50.4**	**54.8**	**21.0**	**0.1**	**21.1**	**57.1**	**184.8**	**Total**
Total Shoreline Miles Spawn				**3.5**					**21.6**			**30.7**			**13.7**	**28.9**	**98.4**	

Table 3. Prince William Sound herring spawn deposition survey biomass estimate, 1989.

Quantity	Symbol	Area					Total
		South East	North-East	North Shore	Naked Island	Montague	
Statute miles of spawn		3.5	21.6	30.7	13.7	28.9	98.4
Kilometers of Spawn		5.6	34.8	49.4	22.0	46.5	158.3
Number of transects possible	*(N)*	17,812	109,927	156,238	69,722	147,078	500,777
Number of transects sampled	*(n)*	6	34	50	20	51	161
No. quadrats sampled in spawn patches	*(sum of m*	10	369	513	225	733	1,850
Proportion of transects sampled	*(f1)*	0.034%	0.031%	0.032%	0.029%	0.035%	0.032%
Proportion of quadrats sampled	*(f2)*	6.325%	6.325%	6.325%	6.325%	6.325%	6.325%
Avg. width of spawn patch ,inc. 0 transects, (m)		8.3	54.3	51.3	56.2	71.9	57.5
Total area of spawn patches (km2)		0.05	1.89	2.53	1.24	3.34	9.05
Avg. of avg. density among transects (1,000/m2)		22.3	182.8	354.2	338.4	209.7	257.9
Avg. total eggs per transect (1,000's)	*(y)*	176	3,577	6,113	7,715	6,309	5,617
Proportion of eggs lost before survey	*(R)*	10%	10%	10%	10%	10%	10%
Total eggs in area (billions)	*(T)*	3	437	1,061	598	1,031	3,130
Avg. herring weight in AWL samples (g)	*(W)*	127	111.0	118.0	117.0	105.7	113
Average weight of females (g)	*(Wf)*	132	115.0	127.0	121.0	111.3	119
Number of Females in AWL Sample		293	200	181	196	544	1414
Number of Fish in AWL Sample		528	433	426	419	1,303	3109
Sex ratio	*(S)*	1.80	2.17	2.35	2.14	2.40	2.27
Fecundity (eggs/average weight of female)	*F(Wf)*	20,112	17,435	19,525	18,466	16,277	17,978
Slope of Fecundity Regression		164.16	166.46	158.20	146.81	189.77	164.16
Intercept of Fecundity Regression		-1557.24	-1708.27	-565.92	702.39	-4850.23	-1557.24
Tonnes of herring per billion eggs	*(B')*	11.38	13.78	14.22	13.54	15.55	14.27
Estimated biomass (tonnes)	*(B)*	39.6	6,022.0	15,093.0	8,095.2	16,031.1	45,281.0
Estimated biomass (short tons)		43.7	6,638.1	16,637.2	8,923.5	17,671.3	49,913.7
Short tons of herring per statute mile		12	307	542	651	611	507
Millions of pounds per statute mile		0.02	0.61	1.1	1.3	1.2	1.0
Distribution by area,							
as percent miles of spawn:		3.6%	22.0%	31.2%	13.9%	29.4%	100.0%
as percent of biomass:		0.1%	13.3%	33.3%	17.9%	35.4%	100.0%

Table 3. (page 2 of 2)

Quantity	Symbol	Area: South East	North-East	North Shore	Naked Island	Montague	Total
Variances of egg counts:							
Among transect variance	*(s12)*	1.9E+05	2.0E+07	4.0E+07	6.3E+07	1.1E+08	6.1E+07
Within transect variance	*(s22)*	2.2E+05	4.1E+07	9.8E+07	1.4E+08	9.4E+08	3.5E+08
Sum of variance of ind. pred. obs.	*(s32)*	1.5E+02	7.1E+03	1.2E+04	5.6E+03	1.5E+04	4.0E+04
Variance of estimated total eggs	*Var(T)*	10	6,971	19,450	15,266	46,404	88,100
Variances from A-W-L-S sampling:							
Variance of average weight	*Var(W)*	1.4848	1.0185	1.3521	1.6134	0.9601	6.4289
Variance of sex ratio	*Var(S)*	0.0049	0.0126	0.0176	0.0124	0.0061	0.0537
MSE from fecundity regression		3.8E+03	3.3E+03	3.9E+03	3.0E+03	3.5E+03	
Mean Weight in Fecundity Sample		120	117	122	126	113	
Sum of x^2 in Fecundity Regression		4.4E+05	1.6E+06	1.6E+06	1.7E+06	1.4E+06	
Number of Fish in Fecundity Sample		407	109	109	101	103	
Variance of est. avg. fecundity	*Var(F(Wf)*	89,636	153,917	221,759	136,708	139,296	741,317
Covariance of avg. wt., fecundity	*Cov(W,F)*						
Variance of B'	*Var(B')*	0.24	0.62	0.78	0.59	0.41	2.64
Precision of esimated biomass:							
Variance of biomass	*Var(B)*	1.6E+03	1.8E+06	5.9E+06	3.7E+06	1.4E+07	2.6E+07
Standard error of B		40	1,333	2,434	1,926	3,790	5,076
Coefficient of variation of B		100%	22%	16%	24%	24%	11%
95% conf. int. width as +/- % of B		196%	43%	32%	47%	46%	22%
Confidence limits on estimated biomass:							
Lower 95% limit, tonnes		(38)	3,410	10,323	4,321	8,604	35,331
Upper 95% limit, tonnes		117	8,634	19,863	11,869	23,459	55,231
Lower 95% limit, short tons		(34)	4,026	11,867	5,149	10,244	39,964
Upper 95% limit, short tons		121	9,250	21,408	12,698	25,099	59,864

Table 4. Prince William Sound herring spawn deposition survey biomass estimate, 1990.

Quantity	Symbol	Area: South East	North-East	North Shore	Naked Island	Montague	Total
Statute miles of spawn		2.6	43.7	18.2	5.4	24.2	94.1
Kilometers of Spawn		4.2	70.3	29.3	8.7	38.9	151.4
Number of transects possible	*(N)*	13,232	222,398	92,623	27,482	123,158	478,893
Number of transects sampled	*(n)*	5	80	31	9	43	168
No. quadrats sampled in spawn patches	*(sum of mi)*	79	1,355	435	74	1,061	3,004
Proportion of transects sampled	*(f1)*	0.038%	0.036%	0.033%	0.033%	0.035%	0.035%
Proportion of quadrats sampled	*(f2)*	6.325%	6.325%	6.325%	6.325%	6.325%	6.325%
Avg. width of spawn patch ,inc. 0 transects, (m)		79.0	84.7	70.2	41.1	123.4	89.4
Total area of spawn patches (km2)		0.33	5.96	2.06	0.36	4.80	13.50
Avg. of avg. density among transects (1,000/m2)		102.4	399.6	479.0	333.9	406.8	403.7
Avg. total eggs per transect (1,000's)	*(y)*	1,610	12,626	12,738	8,316	21,403	14,335
Proportion of eggs lost before survey	*(R)*	10%	10%	10%	10%	10%	10%
Total eggs in area (billions)	*(T)*	24	3,120	1,311	254	2,929	7,637
Avg. herring weight in AWL samples (g)	*(W)*	138	138.0	134.0	138.0	121.0	133
Average weight of females (g)	*(Wf)*	143	143.0	139.0	143.0	127.0	138
Number of Females in AWL Sample		838	838	255	238	188	
Number of Fish in AWL Sample		1,774	1,774	424	423	439	
Sex ratio	*(S)*	2.12	2.12	1.66	1.78	2.34	2.07
Fecundity (eggs/average weight of female)	*F(Wf)*	19,210	19,210	18,636	19,210	16,915	18,636
Slope of Fecundity Regression		143.42	143.42	143.42	143.42	143.42	
Intercept of Fecundity Regression		-1298.93	-1298.93	-1298.93	-1298.93	-1298.93	
Tonnes of herring per billion eggs	*(B')*	15.21	15.21	11.96	12.77	16.70	14.74
Estimated biomass (tonnes)	*(B)*	359.9	47,448.2	15,672.6	3,242.1	48,923.2	115,646.0
Estimated biomass (short tons)		396.8	52,302.6	17,276.0	3,573.8	53,928.5	127,477.7
Short tons of herring per statute mile		153	1197	949	662	2228	1,355
Millions of pounds per statute mile		0.31	2.39	1.9	1.3	4.5	2.7
Distribution by area,							
as percent miles of spawn:		2.8%	46.4%	19.3%	5.7%	25.7%	100.0%
as percent of biomass:		0.3%	41.0%	13.6%	2.8%	42.3%	100.0%

Table 4. (page 2 of 2)

Quantity	Symbol	Area: South East	North-East	North Shore	Naked Island	Montague	Total
Variances of egg counts:							
Among transect variance	*(s12)*	4.4E+06	5.2E+08	2.3E+08	2.0E+08	1.1E+09	5.9E+08
Within transect variance	*(s22)*	2.3E+07	1.1E+09	8.1E+08	2.9E+08	3.7E+09	1.6E+09
Sum of variance of ind. pred. obs.	*(s32)*	1.1E+03	3.5E+04	1.0E+04	1.9E+03	2.5E+04	7.3E+04
Variance of estimated total eggs	*Var(T)*	156	318,599	63,936	17,104	384,797	784,593
Variances from A-W-L-S sampling:							
Variance of average weight	*Var(W)*	1.8557	1.8557	1.2476	1.5981	1.1025	7.6596
Variance of sex ratio	*Var(S)*	0.0028	0.0028	0.0043	0.0058	0.0166	0.0324
MSE from fecundity regression		3.8E+03	3.8E+03	3.8E+03	3.8E+03	3.8E+03	
Mean Weight in Fecundity Sample		144	144	144	144	144	
Sum of x^2 in Fecundity Regression		7.7E+06	7.7E+06	7.7E+06	7.7E+06	7.7E+06	
Number of Fish in Fecundity Sample		353	353	353	353	353	
Variance of est. avg. fecundity	*Var(F(Wt))*	58,447	58,447	98,097	102,116	118,884	435,991
Covariance of avg. wt., fecundity	*Cov(W,F)*						
Variance of B'	*Var(B')*	0.20	0.20	0.27	0.36	0.99	2.03
Precision of esimated biomass:							
Variance of biomass	*Var(B)*	4.5E+04	9.3E+07	1.2E+07	3.5E+06	1.4E+08	2.5E+08
Standard error of B		211	9,662	3,441	1,861	11,938	15,850
Coefficient of variation of B		59%	20%	22%	57%	24%	14%
95% conf. int. width as +/- % of B		115%	40%	43%	113%	48%	27%
Confidence limits on estimated biomass:							
Lower 95% limit, tonnes		(54)	28,512	8,928	(405)	25,525	84,581
Upper 95% limit, tonnes		774	66,385	22,417	6,890	72,322	146,711
Lower 95% limit, short tons		(17)	33,366	10,531	(74)	30,530	96,413
Upper 95% limit, short tons		810	71,239	24,021	7,221	77,327	158,543

Table 5. Prince William Sound herring spawn deposition survey biomass estimate, 1991.

Quantity	Symbol	Area					Total
		South East	North-East	North Shore	Naked Island	Montague	
Statute miles of spawn		3.9	28.5	1.2	0	24.4	58
Kilometers of Spawn		6.3	45.9	1.9	0.0	39.3	93.3
Number of transects possible	*(N)*	19,848	145,042	6,107	0	124,176	295,173
Number of transects sampled	*(n)*	6	41	5	0	51	103
No. quadrats sampled in spawn patches	*(sum of mi)*	59	713	29	0	1,950	2,751
Proportion of transects sampled	*(f1)*	0.030%	0.028%	0.082%		0.041%	0.035%
Proportion of quadrats sampled	*(f2)*	6.325%	6.325%	6.325%		6.325%	6.325%
Avg. width of spawn patch ,inc. 0 transects, (m)		49.2	87.0	29.0		191.2	133.5
Total area of spawn patches (km2)		0.31	3.99	0.06	0.00	7.51	11.86
Avg. of avg. density among transects (1,000/m2)		109.7	385.0	180.2		667.1	498.7
Avg. total eggs per transect (1,000's)	*(y)*	2,084	15,614	2,829		46,454	29,476
Proportion of eggs lost before survey	*(R)*	10%	10%	10%		10%	10%
Total eggs in area (billions)	*(T)*	46	2,516	19		6,409	8,991
Avg. herring weight in AWL samples (g)	*(W)*	139	139.0	145.0		100.0	131
Average weight of females (g)	*(Wf)*	144	144.0	148.0		113.0	137
Number of Females in AWL Sample		1,986	1986	226		1575	
Number of Fish in AWL Sample		3,958	3,958	424		4,077	
Sex ratio	*(S)*	1.99	1.99	1.88		2.59	2.24
Fecundity (eggs/average weight of female)	*F(Wf)*	20,993	20,993	21,343		17,683	20,253
Slope of Fecundity Regression		140.98	140.98	182.74		127.15	
Intercept of Fecundity Regression		691.87	691.87	-5702.07		3315.54	
Tonnes of herring per billion eggs	*(B')*	13.20	13.20	12.75		14.64	14.47
Estimated biomass (tonnes)	*(B)*	606.6	33,205.4	244.7	0.0	93,824.0	127,880.7
Estimated biomass (short tons)		668.6	36,602.7	269.7	0.0	103,423.1	140,964.2
Short tons of herring per statute mile		171	1284	225		4239	2,430
Millions of pounds per statute mile		0.34	2.57	0.4		8.5	4.9
Distribution by area,							
as percent miles of spawn:		6.7%	49.1%	2.1%	0.0%	42.1%	100.0%
as percent of biomass:		0.5%	26.0%	0.2%	0.0%	73.4%	100.0%

Table 5. (page 2 of 2)

Quantity	Symbol	Area: South East	North-East	North Shore	Naked Island	Montague	Total
Variances of egg counts:							
Among transect variance	*(s12)*	6.9E+06	1.4E+09	1.4E+07	.	3.2E+09	2.4E+09
Within transect variance	*(s22)*	1.1E+07	1.2E+09	2.6E+07	.	3.0E+10	1.5E+10
Sum of variance of ind. pred. obs.	*(s32)*	1.9E+03	3.2E+04	1.1E+03	.	9.1E+04	1.3E+05
Variance of estimated total eggs	*Var(T)*	453	695,836	107		980,929	1,677,325
Variances from A-W-L-S sampling:							
Variance of average weight	*Var(W)*	1.2193	0.8317	1.3585		0.4327	3.8422
Variance of sex ratio	*Var(S)*	0.0010	0.0010	0.0073		0.0026	0.0119
MSE from fecundity regression		4.6E+03	4.6E+03	3.9E+03		4.0E+03	
Mean Weight in Fecundity Sample		146	146	152		148	
Sum of x^2 in Fecundity Regression		1.7E+06	1.7E+06	1.6E+06		1.7E+06	
Number of Fish in Fecundity Sample		78	78	66		74	
Variance of est. avg. fecundity	*Var(F(Wf))*	282,704	282,704	295,928		238,902	1,100,236
Covariance of avg. wt., fecundity	*Cov(W,F)*						
Variance of B'	*Var(B')*	0.17	0.16	0.45		0.26	1.04
Precision of esimated biomass:							
Variance of biomass	*Var(B)*	9.8E+04	1.5E+08	2.2E+04		2.7E+08	4.2E+08
Standard error of B		312	12,277	147		16,498	20,568
Coefficient of variation of B		52%	37%	60%		18%	16%
95% conf. int. width as +/- % of B		101%	72%	118%		34%	32%
Confidence limits on estimated biomass:							
Lower 95% limit, tonnes		(6)	9,143	(43)		61,487	87,568
Upper 95% limit, tonnes		1,219	57,268	533		126,161	168,194
Lower 95% limit, short tons		56	12,540	(18)		71,086	100,651
Upper 95% limit, short tons		1,281	60,665	558		135,760	181,277

Table 6. Prince William Sound herring spawn deposition survey biomass estimate, 1992.

Quantity	Symbol	Area					Total
		South East	North-East	North Shore	Naked Island	Montague	
Statute miles of spawn		7.2	32.2	0	0.3	35	74.7
Kilometers of Spawn		11.6	51.8	0.0	0.5	56.3	120.2
Number of transects possible	*(N)*	36,642	163,872	0	1,527	178,122	380,163
Number of transects sampled	*(n)*	17	70	0	4	70	161
No. quadrats sampled in spawn patches	*(sum of mi)*	219	1,004	0	9	1,289	2,521
Proportion of transects sampled	*(f1)*	0.046%	0.043%		0.262%	0.039%	0.098%
Proportion of quadrats sampled	*(f2)*	6.325%	6.325%		6.325%	6.325%	6.325%
Avg. width of spawn patch ,inc. 0 transects, (m)		64.4	71.7		11.3	92.1	78.3
Total area of spawn patches (km2)		0.75	3.71	0.00	0.01	5.19	9.65
Avg. of avg. density among transects (1,000/m2)		400.7	536.9		77.0	647.9	559.4
Avg. total eggs per transect (1,000's)	*(y)*	13,953	18,063		730	19,504	17,825
Proportion of eggs lost before survey	*(R)*	10%	10%		10%	10%	10%
Total eggs in area (billions)	*(T)*	568	3,289	0	1	3,860	7,718
Avg. herring weight in AWL samples (g)	*(W)*	100.3	117.5		115.3	97.9	108
Average weight of females (g)	*(Wf)*	107	123.0		122.0	102.0	114
Number of Females in AWL Sample		143	1020		201	370	
Number of Fish in AWL Sample		444	2,139		445	875	
Sex ratio	*(S)*	3.10	2.10		2.21	2.36	2.32
Fecundity (eggs/average weight of female)	*F(Wf)*	15,631	18,012		17,863	14,887	16,598
Slope of Fecundity Regression		148.80	148.80		148.80	148.80	
Intercept of Fecundity Regression		-290.36	-290.36		-290.36	-290.36	
Tonnes of herring per billion eggs	*(B')*	19.92	13.68		14.29	15.55	15.16
Estimated biomass (tonnes)	*(B)*	11,317.9	44,991.5	0.0	17.7	60,031.3	116,358.4
Estimated biomass (short tons)		12,475.8	49,594.5	0.0	19.5	66,173.1	128,263.0
Short tons of herring per statute mile		1733	1540		65	1891	1,717
Millions of pounds per statute mile		3.47	3.08		0.1	3.8	3.4
Distribution by area,							
as percent miles of spawn:		9.6%	43.1%	0.0%	0.4%	46.9%	100.0%
as percent of biomass:		9.7%	38.7%	0.0%	0.0%	51.6%	100.0%

Table 6. (page 2 of 2)

Quantity	Symbol	Area: South East	North-East	North Shore	Naked Island	Montague	Total
Variances of egg counts:							
Among transect variance	*(s12)*	9.7E+08	8.5E+08		2.1E+06	7.7E+08	8.0E+08
Within transect variance	*(s22)*	1.0E+09	2.1E+09		1.8E+06	4.1E+09	2.8E+09
Sum of variance of ind. pred. obs.	*(s32)*	1.1E+04	4.9E+04		3.1E+02	5.8E+04	1.2E+05
Variance of estimated total eggs	*Var(T)*	76,318	324,971		1	346,794	748,085
Variances from A-W-L-S sampling:							
Variance of average weight	*Var(W)*	2.0270	0.5405		2.3011	0.8960	5.7646
Variance of sex ratio	*Var(S)*	0.0457	0.0023		0.0134	0.0087	0.0701
MSE from fecundity regression		4.2E+03	4.2E+03		4.2E+03	4.2E+03	4.2E+03
Mean Weight in Fecundity Sample		149	149		149	149	149
Sum of x^2 in Fecundity Regression		5.0E+06	5.0E+06		5.0E+06	5.0E+06	5.0E+06
Number of Fish in Fecundity Sample		218	218		218	218	218
Variance of est. avg. fecundity	*Var(F(Wt))*	213,490	102,029		173,695	138,310	627,525
Covariance of avg. wt., fecundity	*Cov(W,F)*						
Variance of B'	*Var(B')*	2.31	0.16		0.70	0.55	3.73
Precision of esimated biomass:							
Variance of biomass	*Var(B)*	3.8E+07	7.7E+07		3.1E+02	1.1E+08	2.3E+08
Standard error of B		6,173	8,785		18	10,651	15,124
Coefficient of variation of B		55%	20%		100%	18%	13%
95% conf. int. width as +/- % of B		107%	38%		196%	35%	25%
Confidence limits on estimated biomass:							
Lower 95% limit, tonnes		(780)	27,772		(17)	39,155	86,716
Upper 95% limit, tonnes		23,416	62,211		52	80,907	146,001
Lower 95% limit, short tons		378	32,375		(15)	45,297	98,621
Upper 95% limit, short tons		24,574	66,814		54	87,049	157,905

Table 7. Mean dates and standard deviation from annual time series observations of daily herring abundance and miles of spawn, Prince William Sound, 1973 - 1992.

	Biomass		Spawn	
Year	Mean Date	SD (days)	Mean Date	SD (days)
1973	-		4/22	11
1974	4/18	3.1	4/22	3.9
1975	-		4/18	5
1976	4/16	1.7	4/17	4.6
1977	4/20	2.2	4/22	6.7
1978	5/03	22.0	4/18	5.2
1979	4/24	12.5	4/22	4.0
1980	4/16	5.5	4/20	7.4
1981	4/16	10.2	4/20	7.4
1982	4/28	4.0	5/01	2.3
1983	4/17	9.5	4/25	8.2
1984	4/16	6.8	4/21	9.1
1985	4/30	2.9	4/29	3.4
1986	4/20	5.3	4/23	6.7
1987	4/16	10.6	4/18	10.4
1988	4/21	4.1	4/22	4.9
1989	4/13	3.7	4/14	3.4
1990	4/13	1.8	4/16	3.2
1991	4/15	7.8	4/22	9.4
1992	4/15	4.9	4/21	5.4
Mean	4/19	9.8	4/21	8.0

Table 8. Mean body weight of female Pacific herring sampled for fecundity analysis, Prince William Sound, 1988-1992.

		Mean Fecundity					
Year		Montague Island	Northeast	Naked Island	North Shore	Southeast	Total
1988	n	65	169		76		310
	Mean	107.8	113.3		130.6		116.4
	STD	41.4	41.2		43.8		42.6
1989	n	103	110	101	101		415
	Mean	112.8	117.4	127.0	121.7		119.6
	STD	28.1	28.9	39.4	33.1		32.9
1990	n	103	86	74	84		347
	Mean	125.9	160.8	148.1	146.7		144.3
	STD	22.8	37.7	34.0	29.6		33.6
1991	n	74	78		66		218
	Mean	148.3	146.3		152.0		148.7
	STD	29.9	30.7		30.3		30.3
1992	n	111	94	122		74	401
	Mean	117.3	130.2	130.5		118.8	124.6
	STD	36.7	44.5	37.4		33.4	38.7
Total	n	456	537	297	327	74	1,691
	Mean	121.9	129.5	133.7	136.3	118.8	129.0
	STD	31.2	37.2	37.2	34.1	33.4	35.9

Table 9. Mean fecundity of female Pacific herring sampled for fecundity analysis, Prince William Sound, 1988-1992.

		Mean Fecundity					
Year		Montague Island	Northeast	Naked Island	North Shore	Southeast	Total
1988	n	65	169		76		310
	Mean	15,175	16,162		19,962		16,887
	STD	8,453	8,221		8,480		8,499
1989	n	103	110	101	101		415
	Mean	16,552	17,931	19,027	18,710		18,045
	STD	6,333	5,984	6,332	6,565		6,349
1990	n	103	86	74	84		347
	Mean	16,463	23,841	18,961	18,637		19,350
	STD	4,655	6,117	6,351	4,964		6,136
1991	n	74	78		66		218
	Mean	22,105	21,237		22,129		21,801
	STD	5,394	6,270		6,517		6,053
1992	n	111	94	122		74	401
	Mean	18,570	21,491	20,701		17,805	19,762
	STD	6,680	8,323	7,159		7,491	7,497
Total	n	456	537	297	327	74	1,691
	Mean	17,728	19,424	19,698	19,672	17,805	18,992
	STD	6,188	7,160	6,677	6,589	7,491	6,933

Table 10. Mean egg weight of female Pacific herring sampled for fecundity analysis, Prince William Sound, 1988-1992.

		Mean Egg Weight (g)					
Year		Montague Island	Northeast	Naked Island	North Shore	Southeast	Total
1988	n	65	169		76		310
	Mean	0.00160	0.00140		0.00140		0.00144
	STD	0.00065	0.00030		0.00023		0.00039
1989	n	103	110	101	101		415
	Mean	0.00145	0.00138	0.00140	0.00142		0.00141
	STD	0.00027	0.00022	0.00023	0.00033		0.00027
1990	n	103	86	74	84		347
	Mean	0.00166	0.00152	0.00180	0.00179		0.00169
	STD	0.00034	0.00025	0.00033	0.00026		0.00032
1991	n	74	78		66		218
	Mean	0.00151	0.00154		0.00158		0.00154
	STD	0.00025	0.00033		0.00023		0.00028
1992	n	111	94	122		74	401
	Mean	0.00132	0.00120	0.00135		0.00122	0.00128
	STD	0.00018	0.00030	0.00017		0.00024	0.00023
Total	n	456	537	297	327	74	1,691
	Mean	0.00150	0.00140	0.00148	0.00154	0.00122	0.00146
	STD	0.00031	0.00028	0.00023	0.00027	0.00024	0.00029

Table 11. Mean gonad weight of female Pacific herring sampled for fecundity analysis, Prince William Sound, 1988-1992.

		Mean Gonad Weight (g)					
Year		Montague Island	Northeast	Naked Island	North Shore	Southeast	Total
1988	n	65	169		76		310
	Mean	22.6	22.5		28.2		23.9
	STD	12.4	12.1		13.2		12.7
1989	n	103	110	101	101		415
	Mean	23.5	24.5	26.9	26.1		25.2
	STD	8.4	8.9	10.8	9.8		9.5
1990	n	103	86	74	84		347
	Mean	26.8	36.6	33.2	33.0		32.1
	STD	7.6	11.6	10.0	8.7		10.1
1991	n	74	78		66		218
	Mean	33.0	31.9		34.2		33.0
	STD	8.9	8.9		8.8		8.9
1992	n	111	94	122		74	401
	Mean	24.6	26.5	27.7		21.5	25.4
	STD	10.0	13.3	10.2		10.4	11.2
Total	n	456	537	297	327	74	1,691
	Mean	25.9	27.2	28.8	30.0	21.5	27.4
	STD	9.3	11.1	10.4	10.1	10.4	10.5

Table 12. Mean gonosomatic index(GSI) of female Pacific herring sampled for fecundity analysis, Prince William Sound, 1988-1992.

		Mean Gonosomatic Index (GSI)					
Year		Montague Island	Northeast	Naked Island	North Shore	Southeast	Total
1988	n	65	169		76		310
	Mean	21%	20%		22%		21%
	STD	30%	29%		30%		30%
1989	n	103	110	101	101		415
	Mean	21%	21%	21%	21%		21%
	STD	30%	31%	27%	30%		29%
1990	n	103	86	74	84		347
	Mean	21%	23%	22%	23%		22%
	STD	33%	31%	29%	29%		30%
1991	n	74	78		66		218
	Mean	22%	22%		23%		22%
	STD	30%	29%		29%		29%
1992	n	111	94	122		74	401
	Mean	21%	20%	21%		18%	20%
	STD	27%	30%	27%		31%	29%
Total	n	456	537	297	327	74	1,691
	Mean	21%	21%	21%	22%	18%	21%
	STD	30%	30%	28%	30%	31%	29%

Table 13. SAS output for analysis of variance (ANOVA) of log(fecundity), log(egg weight) and log(gonad weight) of Pacific herring in Prince William Sound, Alaska.

Analysis of variance of Log(Fecundity)

Dependent Variable: LOGFEC

Source	DF	Sum of Squares	Mean Square	F Value	Pr > F
Model	17	32.42917756	1.90759868	14.00	0.0001
Error	1673	227.99135347	0.13627696		
Corrected Total	1690	260.42053103			

R-Square	C.V.	Root MSE	LOGFEC Mean
0.124526	3.775090	0.369157	9.77876146

Source	DF	Type III SS	Mean Square	F Value	Pr > F
YEAR	4	16.05861434	4.01465359	29.46	0.0001
AREA	4	6.97388432	1.74347108	12.79	0.0001
YEAR*AREA	9	8.49987846	0.94443094	6.93	0.0001

Contrast	DF	Contrast SS	Mean Square	F Value	Pr > F
NE vs rest:90 vs 89	1	3.07972003	3.07972003	22.60	0.0001

Analysis of variance of Log(Egg Weight)

Dependent Variable: LOGEW

Source	DF	Sum of Squares	Mean Square	F Value	Pr > F
Model	17	19.20966225	1.12998013	28.74	0.0001
Error	1670	65.66468850	0.03932017		
Corrected Total	1687	84.87435075			

R-Square	C.V.	Root MSE	LOGEW Mean
0.226331	3.025501	0.198293	-6.5540608

Source	DF	Type III SS	Mean Square	F Value	Pr > F
YEAR	4	13.25592225	3.31398056	84.28	0.0001
AREA	4	2.04299994	0.51074998	12.99	0.0001
YEAR*AREA	9	1.71329430	0.19036603	4.84	0.0001

Contrast	DF	Contrast SS	Mean Square	F Value	Pr > F
NE vs rest:90 vs 89	1	0.41736373	0.41736373	10.61	0.0011

Analysis of variance of Log(Gonad Weight)

Dependent Variable: LOGGW

Source	DF	Sum of Squares	Mean Square	F Value	Pr > F
Model	17	53.19547174	3.12914540	19.70	0.0001
Error	1670	265.31701407	0.15887246		
Corrected Total	1687	318.51248581			

R-Square	C.V.	Root MSE	LOGGW Mean
0.167012	12.36639	0.398588	3.22315569

Source	DF	Type III SS	Mean Square	F Value	Pr > F
YEAR	4	33.50501048	8.37625262	52.72	0.0001
AREA	4	8.00717065	2.00179266	12.60	0.0001
YEAR*AREA	9	4.82106565	0.53567396	3.37	0.0004

Contrast	DF	Contrast SS	Mean Square	F Value	Pr > F
NE vs rest:90 vs 89	1	1.22705405	1.22705405	7.72	0.0055

Table 14. Coefficient of determination (R^2) values for linear regressions using either body weight, gonad weight, gonosomatic Index (GSI), fecundity, or egg weight for Pacific herring in Prince William Sound from 1988-1992 as dependent variables and mean sea surface temperature anomolies (SSTA's) at three grid points in the western Gulf of Alaska (55°N, 160°W; 60°N, 150°W; 60°N, 155°W) as the independent variable. Mean sea surface temperature anomolies were grouped into 3 month periods prior to April, the month during which herring usually spawn within Prince William Sound.

3 Month Periods Prior to Spawning	R^2 Values for Linear Regresions				
	Body Weight	Gonad Weight	GSI	Fecundity	Egg Weight
Jan, Feb, Mar (13-15 Months Before)	0.827	0.777	0.572	0.657	0.312
Apr, May, Jun (10-12 Months Before	0.168	0.196	0.228	0.021	0.317
Jul, Aug, Sep (7-9 Months Before)	0.438	0.544	0.808	0.039	0.909
Oct, Nov, Dec (4-6 Months Before	0.333	0.414	0.618	0.004	0.883
Jan, Feb, Mar (1-3 Months Before)	0.105	0.164	0.385	0.010	0.274
Apr, May, Jun (During Spawning)	0.000	0.001	0.000	0.027	0.012

Table 15. First day of observed spawn, eyed eggs (number of days from spawning to visible eyes in embryos), and hatch (incubation period) for Pacific herring at egg loss transects in Prince William Sound, 1990-1991.

Transect	Location	Area	First Day of Observed Spawn	First Day of Eyed Eggs	Number of Days to Eye		First Day of Hatched Eggs	Number Days to Hatch	
					Mean	SD		Mean	SD
1990									
C2	Fairmont Bay	North Shore	16-Apr	06-May	20		10-May	24	
C3	Fairmont Bay	North Shore	16-Apr	06-May	20		10-May	24	
C6	Fairmont Bay	North Shore	16-Apr	06-May	20		10-May	24	
North Shore Area			16-Apr	06-May	20.0	0.0	10-May	24.0	0.0
O18	Rocky Bay	Montague Island	18-Apr	03-May	15		09-May	21	
O20	Rocky Bay	Montague Island	18-Apr	03-May	15		07-May	19	
O21	Rocky Bay	Montague Island	16-Apr	03-May	17		09-May	23	
Montague Island Area			17-Apr	03-May	15.7	1.2	08-May	21.0	2.0
O22	West Peak Island	Naked Island	13-Apr	03-May	20		11-May	28	
O23	Cabin Bay	Naked Island	16-Apr	04-May	18		11-May	25	
O24	McPherson Bay	Naked Island	17-Apr	03-May	16		11-May	24	
Naked Island Area			15-Apr	03-May	18.0	2.0	11-May	25.7	2.1
1990 - All Areas Combined			16-Apr	04-May	17.9	2.2	09-May	23.6	2.5
1991									
C12	Galena Bay	Northeast	21-Apr	08-May	17		13-May	22	
C21	Picnic Cove	Northeast	22-Apr	08-May	16		13-May	21	
Northeast Area			21-Apr	08-May	16.5	0.7	13-May	21.5	0.7
C15	Fairmont Bay	North Shore	23-Apr	10-May	17		18-May	25	
North Shore Area			23-Apr	10-May	17.0		18-May	25.0	
O25	Rocky Bay	Montague Island	20-Apr	10-May	20		16-May	26	
O26	Graveyard Point	Montague Island	25-Apr	12-May	17		16-May	21	
O28	Clam Digger's Cove	Montague Island	23-Apr	12-May	19		17-May	24	
Montague Island Area			22-Apr	11-May	18.7	1.5	16-May	23.7	2.5
1991 - All Areas Combined			22-Apr	09-May	17.4	1.5	15-May	23.4	2.1
1990 & 1991 Combined			19-Apr	07-May	17.6	0.4	12-May	23.5	0.1

Table 16. SAS output from analysis of covariance (ANCOVA) model between egg density estimates by SCUBA divers and laboratory enumerated egg densities in 1990. The estimates from the ANCOVA model were used to correct the estimates of egg densities by SCUBA divers at egg loss transect sampling locations in Prince William Sound, Alaska, 1990-1991.

General Linear Models Procedure
Class Level Information

Class	Levels	Values
DIVER	3	DC DNE SW
VEGGROUP	4	1 2 3 4

Number of observations in data set = 402

Dependent Variable: LOGLAB

Source	DF	Sum of Squares	Mean Square	F Value	Pr > F
Model	10	6304.336698	630.433670	859.63	0.0001
Error	392	287.484663	0.733379		
Uncorrected Total	402	6591.821361			

R-Square	C.V.	Root MSE	LOGLAB Mean
0.823113	24.36401	0.856376	3.51492039

Source	DF	Type III SS	Mean Square	F Value	Pr > F
VEGGROUP	3	5.795780	1.931927	2.63	0.0496
DIVER	2	10.192215	5.096108	6.95	0.0011
LOGDIV(VEGGROUP)	4	1199.332629	299.833157	408.84	0.0001

Parameter		Estimate		T for H0: Parameter=0	Pr > |T|	Std Error of Estimate
EEL-DC		0.27807286		1.16	0.2476	0.24012583
EEL-DNE		-0.19119274		-0.72	0.4732	0.26630415
EEL-SW		0.10062086		0.41	0.6825	0.24577798
HRK-DC		0.35374255		1.51	0.1307	0.23358736
HRK-DNE		-0.11552305		-0.48	0.6341	0.24254147
HRK-SW		0.17629055		0.72	0.4723	0.24501490
FUC-DC		0.86428996		6.75	0.0001	0.12799086
FUC-DNE		0.39502436		2.30	0.0222	0.17210096
FUC-SW		0.68683797		4.37	0.0001	0.15732342
LBK-DC		0.82456663		5.67	0.0001	0.14538173
LBK-DNE		0.35530103		2.24	0.0258	0.15875382
LBK-SW		0.64711464		4.11	0.0001	0.15730113
VEGGROUP	1	0.100620865	B	0.41	0.6825	0.24577798
	2	0.176290554	B	0.72	0.4723	0.24501490
	3	0.686837968	B	4.37	0.0001	0.15732342
	4	0.647114639	B	4.11	0.0001	0.15730113
DIVER	DC	0.177451993	B	1.65	0.0989	0.10728898
	DNE	-0.291813605	B	-2.26	0.0244	0.12917094
	SW	0.000000000	B	.	.	.
LOGDIV(VEGGROUP)	1	1.022646620		14.00	0.0001	0.07307021
	2	1.022355649		17.90	0.0001	0.05712930
	3	0.924141729		22.80	0.0001	0.04052720
	4	0.910184741		25.38	0.0001	0.03585574

NOTE: The X'X matrix has been found to be singular and a generalized inverse was used to solve the normal equations. Estimates followed by the letter 'B' are biased, and are not unique estimators of the parameters.

Table 17. SAS output from analysis of covariance (ANCOVA) model to estimate the loss of Pacific herring eggs in Prince William Sound, Alaska, 1990.

General Linear Models Procedure
Class Level Information

Class	Levels	Values
TRANS	9	C2 C3 C6 O18 O20 O21 O22 O23 O24
DEPTH	6	0 1 5 -5 -15 -30

Number of observations in data set = 1881

NOTE: Due to missing values, only 1879 observations can be used in this analysis.

Dependent Variable: LOGADJ

Source	DF	Sum of Squares	Mean Square	F Value	Pr > F
Model	84	4633.211751	55.157283	40.17	0.0
Error	1794	2463.122913	1.372978		
Corrected Total	1878	7096.334664			

R-Square	C.V.	Root MSE	LOGADJ Mean
0.652902	32.29600	1.171742	3.62813203

Source	DF	Type III SS	Mean Square	F Value	Pr > F
DAYS	1	42.0914290	42.0914290	30.66	0.0001
TRANS	8	301.1196175	37.6399522	27.41	0.0001
DAYS*TRANS	8	41.8190901	5.2273863	3.81	0.0002
DEPTH	5	27.6537296	5.5307459	4.03	0.0012
DAYS*DEPTH	5	27.6226592	5.5245318	4.02	0.0012
TRANS*DEPTH	27	185.1320684	6.8567433	4.99	0.0001
DAYS*TRANS*DEPTH	27	110.4596082	4.0910966	2.98	0.0001

Table 18. SAS output from analysis of covariance (ANCOVA) model to estimate the loss of Pacific herring eggs in Prince William Sound, Alaska, 1991.

General Linear Models Procedure
Class Level Information

Class	Levels	Values
TRANS	6	C12 C15 C21 O25 O26 O28
DEPTH	5	0 1 5 -5 -15

Number of observations in data set = 587

NOTE: Due to missing values, only 584 observations can be used in this analysis.

Dependent Variable: LOGADJ

Source	DF	Sum of Squares	Mean Square	F Value	Pr > F
Model	53	754.2634186	14.2313853	32.21	0.0001
Error	530	234.1913587	0.4418705		
Corrected Total	583	988.4547773			

R-Square	C.V.	Root MSE	LOGADJ Mean
0.763073	16.73702	0.664733	3.97163519

Source	DF	Type III SS	Mean Square	F Value	Pr > F
DAYS	1	3.37338169	3.37338169	7.63	0.0059
TRANS	5	13.38116413	2.67623283	6.06	0.0001
DAYS*TRANS	5	27.64332481	5.52866496	12.51	0.0001
DEPTH	4	10.94749546	2.73687387	6.19	0.0001
DAYS*DEPTH	4	12.44724467	3.11181117	7.04	0.0001
TRANS*DEPTH	17	27.72887603	1.63111035	3.69	0.0001
DAYS*TRANS*DEPTH	17	15.81985155	0.93057950	2.11	0.0060

Table 19. Estimates of the average daily percent loss (negative percents) or gain (positive percents) during the incubation period in Prince William Sound, Alaska, 1990-1991.

Transec	Location	Area	Daily Percent Loss or Gain							
			Depth						All Depths Average	Excluding 1.65 m Average
			5 ft 1.65 m	1 ft 0.33 m	0 ft 0.00 m	-5 ft -1.65 m	-15 ft -4.95 m	-30 ft -9.90 m		
1990										
C2	Fairmont Bay	North Shore		-12.5%	-2.2%	2.5%	5.9%	3.2%	-0.6%	-0.6%
C3	Fairmont Bay	North Shore	-4.1%	3.2%	-0.2%	0.1%	-1.6%		-0.5%	0.4%
C6	Fairmont Bay	North Shore	7.0%	-1.5%	-5.9%	3.1%	3.5%	2.0%	1.4%	0.2%
North Shore Area			1.4%	-3.6%	-2.8%	1.9%	2.6%	2.6%	0.2%	0.1%
O18	Rocky Bay	Montague Island	-27.2%	-2.3%	-5.8%	-16.6%	0.0%		-10.4%	-6.2%
O20	Rocky Bay	Montague Island		1.8%	-32.7%	-8.0%	-8.4%	-9.9%	-11.4%	-11.4%
O21	Rocky Bay	Montague Island	-10.7%	-3.0%	0.2%	-8.6%	-9.9%		-6.4%	-5.3%
Montague Island Area			-19.0%	-1.2%	-12.8%	-11.1%	-6.1%	-9.9%	-9.4%	-8.2%
O22	West Peak Island	Naked Island	-34.9%	-7.3%	-8.4%	-1.8%	-1.6%		-10.8%	-4.8%
O23	Cabin Bay	Naked Island	-19.3%	4.3%	-9.9%	2.8%	-12.4%		-6.9%	-3.8%
O24	McPherson Bay	Naked Island	-4.0%	3.9%	-5.9%	-2.4%	-7.5%		-3.2%	-3.0%
Naked Island Area			-19.4%	0.3%	-8.1%	-0.5%	-7.2%		-7.0%	-3.9%
1990 - All Areas Combined			-12.3%	-1.5%	-7.9%	-3.2%	-3.6%	-3.7%	-5.4%	-4.0%
1991										
C12	Galena Bay	Northeast	-7.4%	-0.3%	1.5%	1.5%	1.5%	-1.8%	-0.8%	0.5%
C21	Picnic Cove	Northeast	-26.2%	-10.0%	-5.3%	-5.3%	-5.3%	0.9%	-8.5%	-5.0%
Northeast Area			-16.8%	-5.1%	-1.9%	-1.9%	-1.9%	-0.4%	-4.7%	-2.3%
C15	Fairmont Bay	North Shore	0.0%	-2.7%	-5.2%	-5.2%	-5.2%	-2.5%	-3.5%	-4.2%
North Shore Area			0.0%	-2.7%	-5.2%	-5.2%	-5.2%	-2.5%	-3.5%	-4.2%
O25	Rocky Bay	Montague Island	0.0%	3.9%	8.7%	8.7%	8.7%	-2.6%	4.6%	5.5%
O26	Graveyard Point	Montague Island	-53.3%	4.9%	5.1%	5.1%	5.1%	7.1%	-4.3%	5.5%
O28	Clam Digger's Cove	Montague Island	-2.9%	6.3%	8.7%	8.7%	8.7%	0.9%	5.1%	6.7%
Montague Island Area			-18.7%	5.0%	7.5%	7.5%	7.5%	1.8%	1.8%	5.9%
1991 - All Areas Combined			-11.8%	-0.9%	0.1%	0.1%	0.1%	-0.4%	-2.1%	-0.2%
1990 & 1991 Combined			-12.1%	-1.2%	-3.9%	-1.5%	-1.7%	-2.0%	-3.8%	-2.1%

Table 20. Mean level of aromatic hydrocarbons, phytane, and estimated oil concentration in mussel samples collected for Fish/Shellfish Study 11, 1989. Levels are in ng/g wet weight (ppb).

Location	Transect	Depth	Date	Number of Samples Analyzed	Aromatic Hydrocarbons Mean	Aromatic Hydrocarbons se	Napthalenes Mean	Napthalenes se	Phenanthrenes Mean	Phenanthrenes se	Dibenzothiophenes Mean	Dibenzothiophenes se	Chrysenes Mean	Chrysenes se	Fluorenes Mean	Fluorenes se	Phytane Mean	Phytane se	Oil Concentration Mean	Oil Concentration se
1989																				
All Sites Combined				75	656.0	81.3	83.3	5.4	269.9	34.8	146.0	24.7	45.0	6.8	80.3	10.0	65.8	9.0	1,322.7	165.9
Control Sites Only	C = Control			15	104.7	13.5	56.7	7.7	27.4	4.1	5.9	1.2	1.1	0.2	2.5	1.1	4.2	1.3	29.4	4.7
Oiled Sites Only	O = Oiled			59	806.9	94.1	91.3	6.2	336.1	40.0	184.1	29.5	56.9	7.9	101.4	11.3	80.5	10.5	1,646.1	186.8
Fairmont Bay	C1	-2.0	28-Apr-89	3	37.4	10.5	22.3	8.2	4.2	0.8	0.6	0.1	0.6	0.1	2.6	1.1	0.0	0.0	28.9	11.0
Fairmont Bay	C2	-1.0	28-Apr-89	3	141.3	41.2	74.3	19.6	37.5	7.8	10.4	3.6	1.2	0.5	6.1	5.3	4.4	3.1	35.4	17.7
Fairmont Bay	C3	-2.0	28-Apr-89	3	130.2	8.9	69.8	7.9	43.5	3.5	6.1	2.4	1.1	0.3	0.7	0.1	10.6	2.3	37.8	8.7
Fairmont Bay	C4	-2.0	28-Apr-89	3	109.7	27.4	63.4	22.8	29.2	3.0	6.0	0.9	1.0	0.2	1.3	0.3	6.0	1.7	19.8	1.9
Fairmont Bay	C5	-2.0	28-Apr-89	3	105.1	21.9	53.6	14.4	22.7	6.0	6.2	1.9	1.6	1.0	1.9	1.3	0.0	0.0	25.2	11.8
Naked Island	O1	-1.0	02-May-89	3	701.1	92.9	76.7	14.6	308.4	40.3	142.7	23.0	58.6	5.0	85.6	24.1	75.3	3.7	1,260.2	429.5
Naked Island	O2	-2.0	02-May-89	3	407.7	87.2	76.0	5.4	156.9	37.8	94.0	20.1	22.3	10.3	36.8	19.7	61.6	3.3	756.4	300.4
Naked Island	O3	-2.0	02-May-89	3	2,532.8	747.6	132.9	36.7	1,032.0	352.7	802.4	230.1	201.8	61.2	256.8	69.0	285.0	101.7	4,332.4	1,160.2
Naked Island	O4	-2.0	02-May-89	3	247.2	52.8	76.5	6.3	97.0	35.1	48.2	16.9	8.2	3.8	1.1	0.1	45.7	1.5	196.4	81.7
Naked Island	O5	-2.0	02-May-89	3	311.8	22.4	42.9	9.7	143.3	24.0	50.0	6.1	11.8	3.2	50.3	2.8	31.0	2.7	787.1	53.5
Naked Island	O6	-2.0	02-May-89	3	184.4	25.8	41.1	8.3	71.2	2.1	27.0	3.1	8.0	3.1	26.3	13.0	17.0	2.5	436.7	140.3
Naked Island	O7	-1.0	02-May-89	3	218.9	45.0	40.2	9.5	91.8	10.5	39.7	7.7	10.2	4.1	24.7	17.1	26.5	5.4	293.2	217.2
Naked Island	O8	-1.0	02-May-89	3	244.6	17.9	69.3	27.5	94.3	15.3	38.5	6.9	5.8	2.7	25.7	13.7	23.9	1.8	424.8	179.4
Naked Island	O9	-1.0	02-May-89	3	471.1	101.3	67.8	13.9	204.6	45.1	84.3	18.9	38.3	9.8	57.5	15.7	44.6	7.4	877.9	301.7
Naked Island	O10	-1.0	02-May-89	3	497.3	36.9	73.8	29.4	216.0	3.4	75.4	1.4	32.7	5.6	65.4	3.0	43.6	5.8	1,163.8	57.4
Naked Island	O11	-2.0	03-May-89	3	848.5	42.1	117.2	4.0	354.2	23.7	165.4	6.6	58.9	7.4	114.3	21.5	89.4	7.5	1,880.1	316.3
Naked Island	O12	-2.0	03-May-89	3	1,203.5	106.4	199.9	36.7	475.0	27.4	215.5	15.5	76.5	12.7	180.6	8.2	93.3	10.0	2,937.0	126.9
Naked Island	O13	-2.0	03-May-89	3	1,104.7	71.2	129.5	20.3	481.9	23.9	216.8	16.0	78.4	9.9	147.1	21.5	159.8	6.2	2,374.6	288.8
Naked Island	O14	-2.0	03-May-89	3	763.7	123.4	79.1	6.6	330.0	66.9	140.4	19.0	62.4	16.3	110.9	16.1	73.4	5.9	1,907.2	347.3
Storey Island	O15	-2.0	03-May-89	3	1,234.7	85.7	95.9	3.6	510.9	50.8	274.5	22.9	83.9	7.5	221.3	0.4	81.9	1.9	3,225.6	121.9
Storey Island	O16	-2.0	03-May-89	3	454.3	25.1	122.1	12.3	153.1	11.5	56.4	4.6	18.2	1.4	81.3	15.4	20.2	4.9	1,226.7	171.1
Rocky Bay	O17A	0.0	04-May-89	2	1,743.7	169.8	118.1	2.9	811.9	82.5	378.6	40.9	146.0	15.5	210.1	20.8	165.6	10.6	2,532.3	1,519.8
Rocky Bay	O17B	-2.0	04-May-89	3	2,283.7	152.0	142.7	6.2	903.9	134.8	713.7	33.0	179.3	14.2	257.7	23.9	229.0	26.9	4,366.0	521.1
Rocky Bay	O18	-1.0	04-May-89	3	115.2	10.8	41.1	6.2	37.4	1.6	27.8	2.5	1.3	0.1	0.6	0.1	0.0	0.0	24.7	2.0
Rocky Bay	O19	-1.0	04-May-89	3	882.2	38.7	91.3	6.1	407.2	39.1	155.6	3.0	65.5	2.6	110.2	5.7	71.2	5.0	1,917.9	13.1

Table 21. Mean level of aromatic hydrocarbons, phytane, and estimated oil concentration in mussel samples collected for Fish/Shellfish Study 11, 1990. Levels are in ng/g wet weight (ppb).

Location	Transect	Depth	Date	Number of Samples Analyzed	Aromatic Hydrocarbons Mean	Aromatic Hydrocarbons se	Napthalenes Mean	Napthalenes se	Phenanthrenes Mean	Phenanthrenes se	Dibenzothiophenes Mean	Dibenzothiophenes se	Chrysenes Mean	Chrysenes se	Fluorenes Mean	Fluorenes se	Phytane Mean	Phytane se	Oil Concentration Mean	Oil Concentration se
1990																				
All Sites Combined				33	96.4	30.9	29.3	4.0	32.4	17.1	6.7	3.6	14.2	8.3	0.7	0.1	11.3	6.6	141.1	76.0
Control Sites Only	C = Control			9	23.4	2.3	13.7	1.7	1.9	0.1	0.3	0.1	0.4	0.0	0.8	0.1	0.0	0.0	9.5	0.9
Oiled Sites Only	O = Oiled			24	123.8	41.4	35.2	5.0	43.8	23.3	9.1	4.9	19.4	11.2	0.7	0.1	15.5	8.9	190.4	103.3
Fairmont Bay	C2	-1.4	21-Apr-89	3	23.3	4.2	12.6	2.3	2.1	0.2	0.4	0.2	0.4	0.1	0.9	0.1	0.0	0.0	9.4	1.1
Fairmont Bay	C3	-1.4	21-Apr-89	3	24.8	6.0	15.2	5.2	1.9	0.2	0.4	0.1	0.4	0.1	0.7	0.1	0.0	0.0	8.4	1.4
Fairmont Bay	C6	-1.4	21-Apr-89	3	22.1	2.3	13.5	1.4	1.5	0.2	0.2	0.0	0.3	0.1	0.9	0.3	0.0	0.0	10.7	2.5
Rocky Bay	O20	0.0	29-Apr-89	3	29.5	7.8	10.2	1.5	8.1	5.0	0.5	0.1	1.1	0.5	0.8	0.1	10.2	1.8	11.6	3.0
Peak Island	O22	-1.4	28-Apr-89	5	63.1	12.8	51.1	11.3	2.7	0.3	0.3	0.1	1.2	0.1	0.5	0.1	9.0	9.0	14.0	4.9
Cabin Bay	O23	-1.4	28-Apr-89	7	50.0	9.5	38.9	8.7	2.7	0.5	0.4	0.0	0.5	0.0	0.7	0.1	0.0	0.0	11.3	1.5
McPherson Bay	O24	-1.4	28-Apr-89	6	57.9	9.1	41.8	9.6	3.4	0.6	0.5	0.1	2.2	0.7	0.7	0.2	0.0	0.0	11.2	1.5
Smith Island	O30	-1.0	10-May-89	3	623.3	113.1	11.9	4.4	324.5	68.9	69.8	11.4	146.7	48.2	0.7	0.2	98.8	55.2	1,439.8	299.1

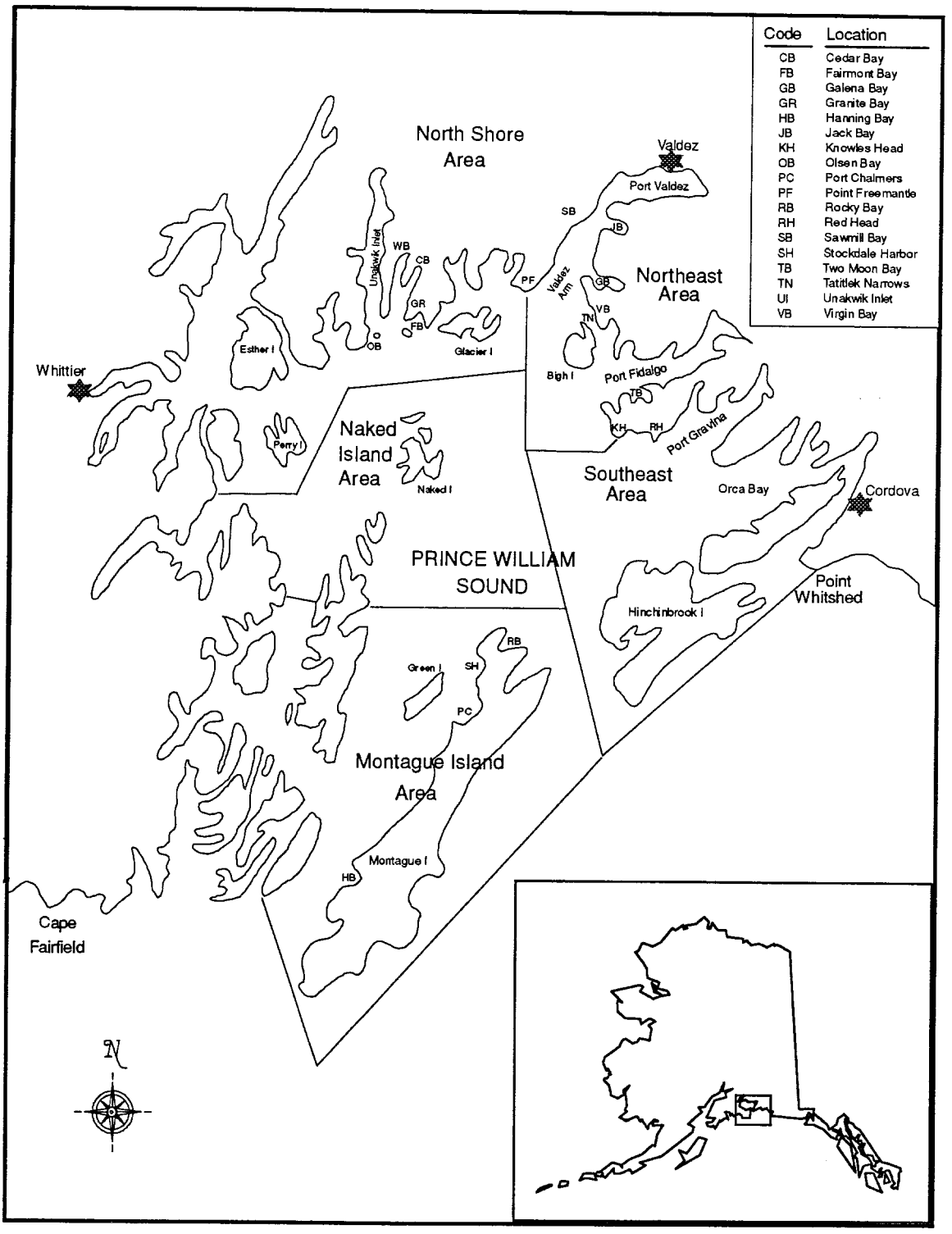


Figure 1. Delineation of the five major herring spawning areas in Prince William Sound; the Southeast, Northeast, North Shore, Naked Island and Montague Island Areas.

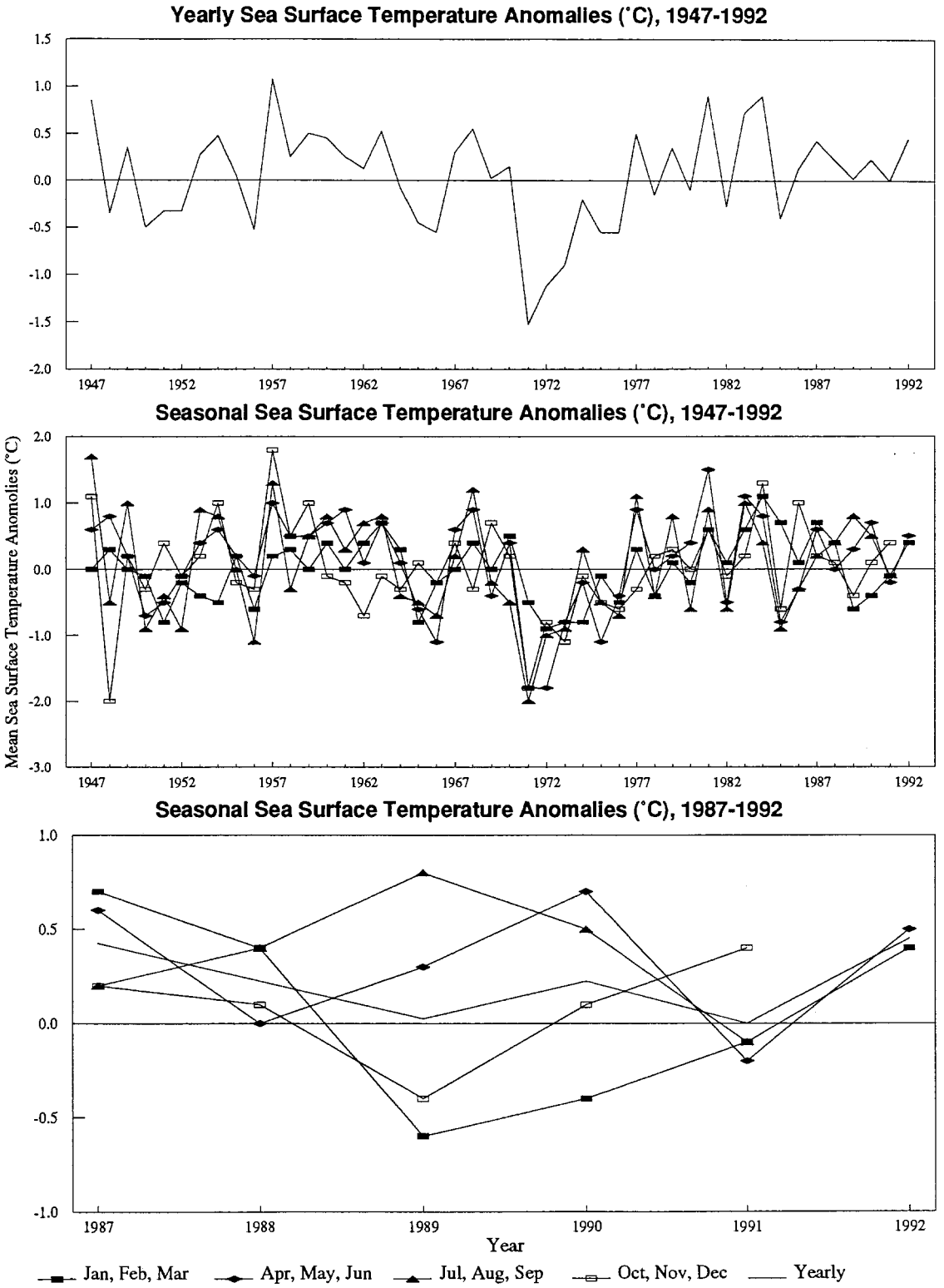


Figure 2. Mean sea surface temperature anomalies (° C) at three grid points in the western Gulf of Alaska (55N, 160W; 60N, 150W; 60N, 155W). Sea surface temperature data was collected by ships of opportunity and more recently by satellite infrared sensors. The data was compiled by Scripps Institute of Oceanography, LaJolla, CA and provided by Mark Willette of the Alaska Department of Fish and Game, Cordova, AK.

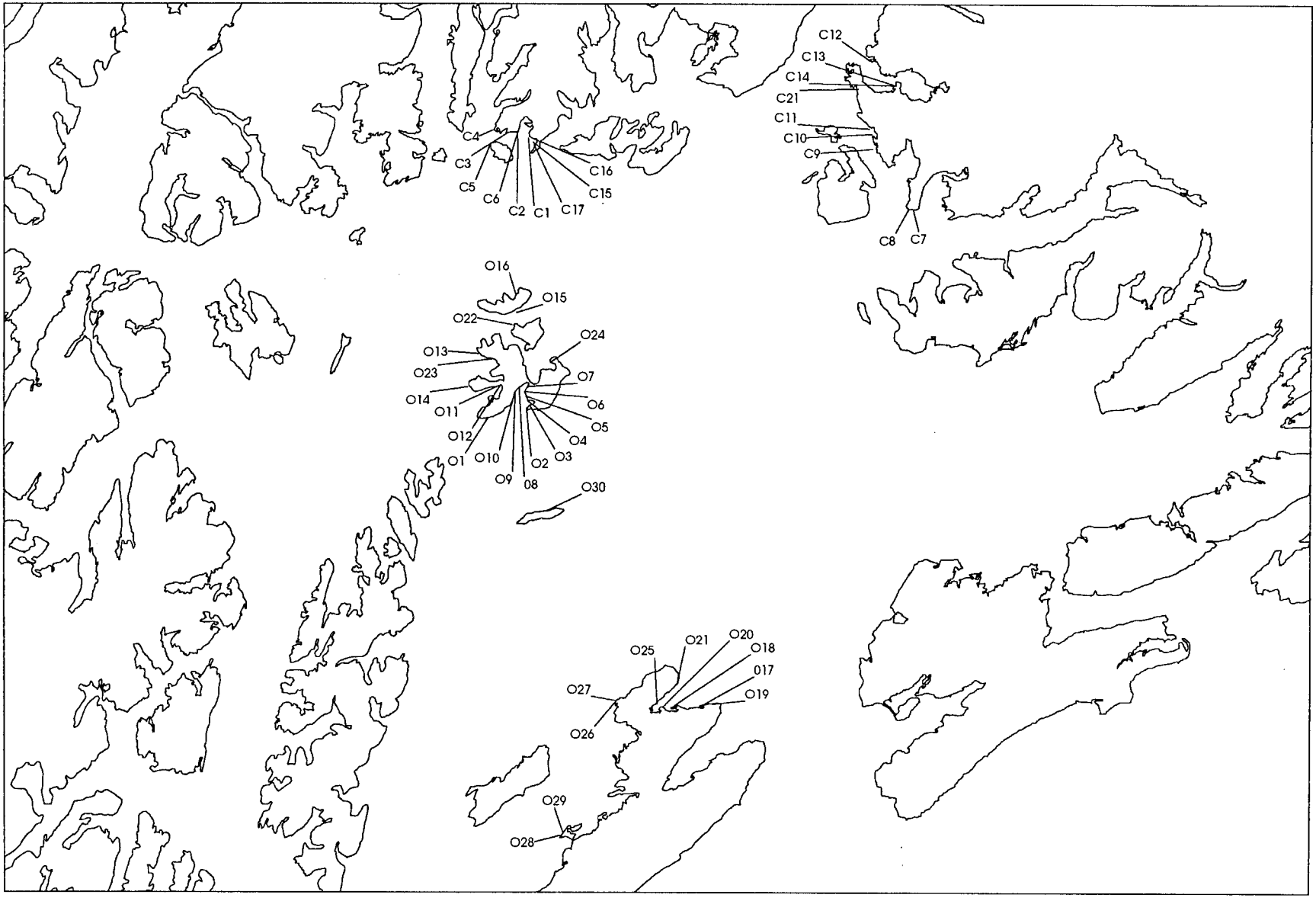


Figure 3a. Sites used for the injury to Prince William Sound herring studies, 1989-1991.

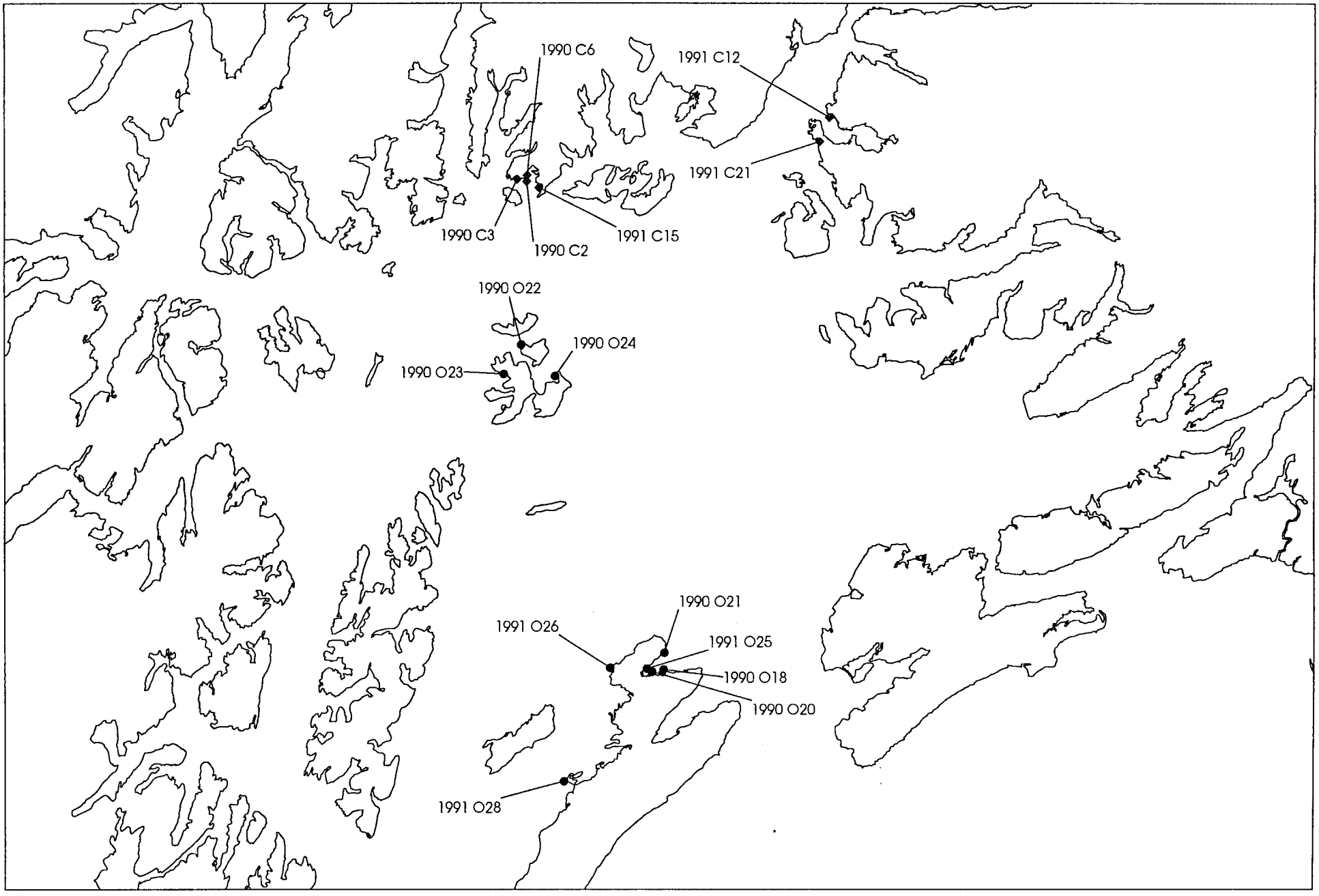


Figure 3b. Injury to Prince William Sound herring study sites for egg loss in spawning areas, 1990-1991.

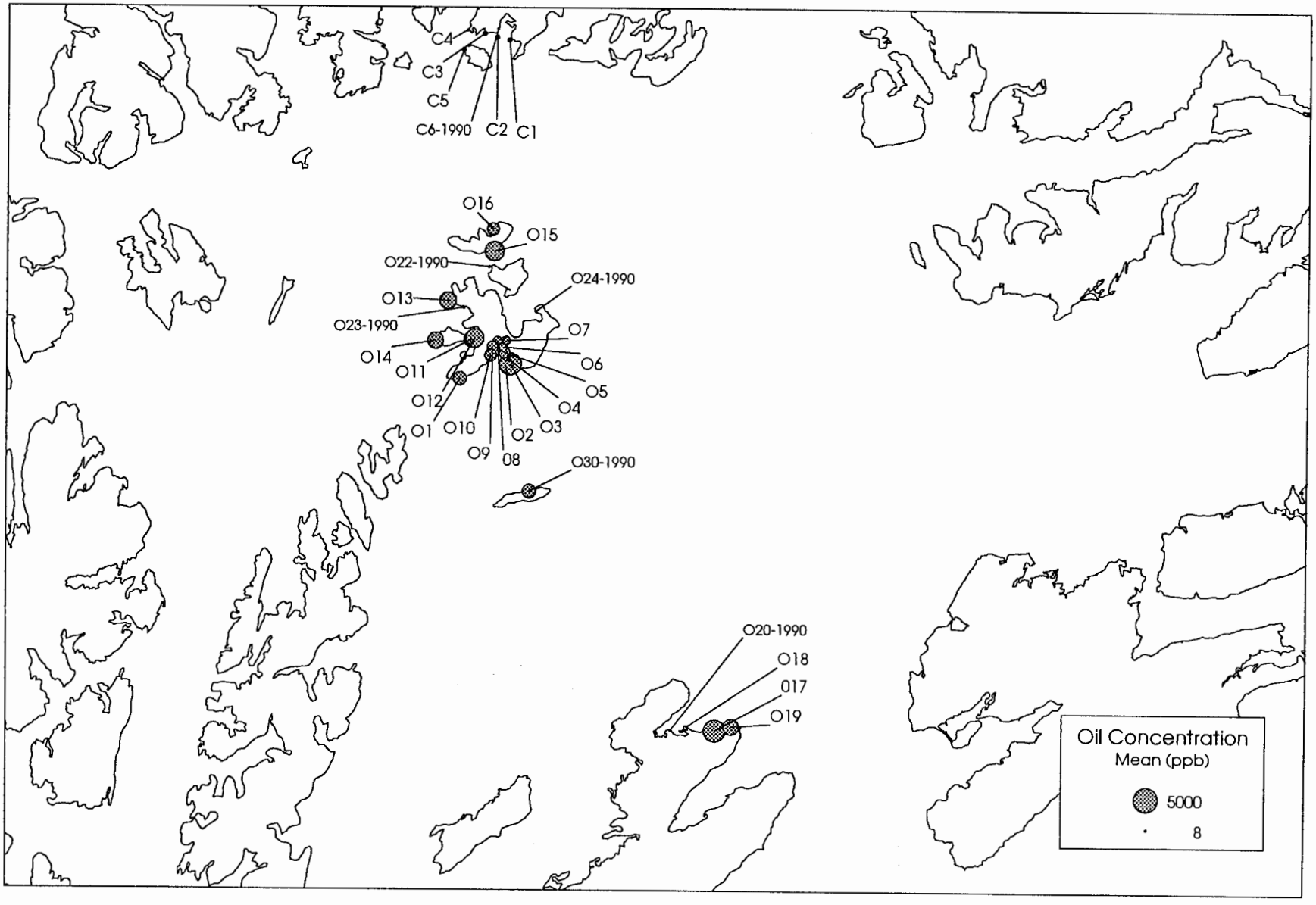
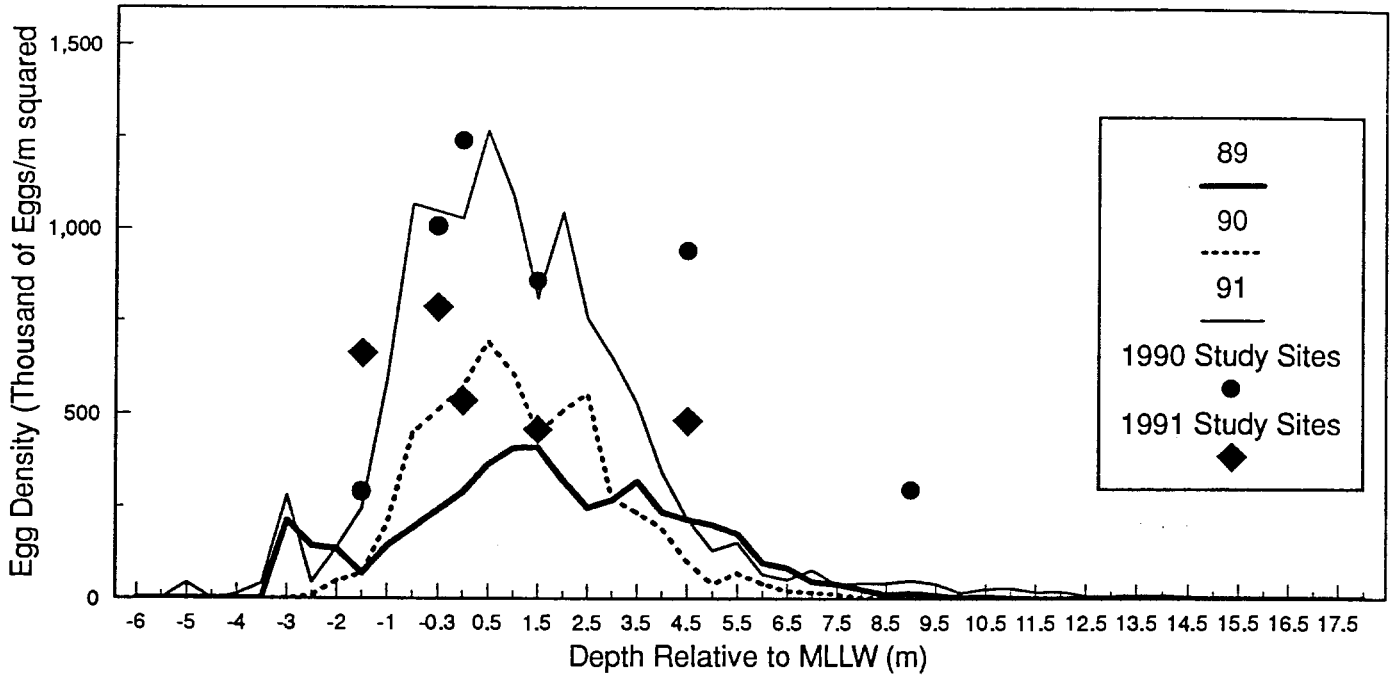


Figure 3c. Prince William Sound herring mussel tissue collection sites and relative petroleum hydrocarbon levels (ppb mean oil concentration), 1989-1990.

Oiled Areas (Montague Island and Naked Island)



Unoiled Areas (Northeast and North Shore)

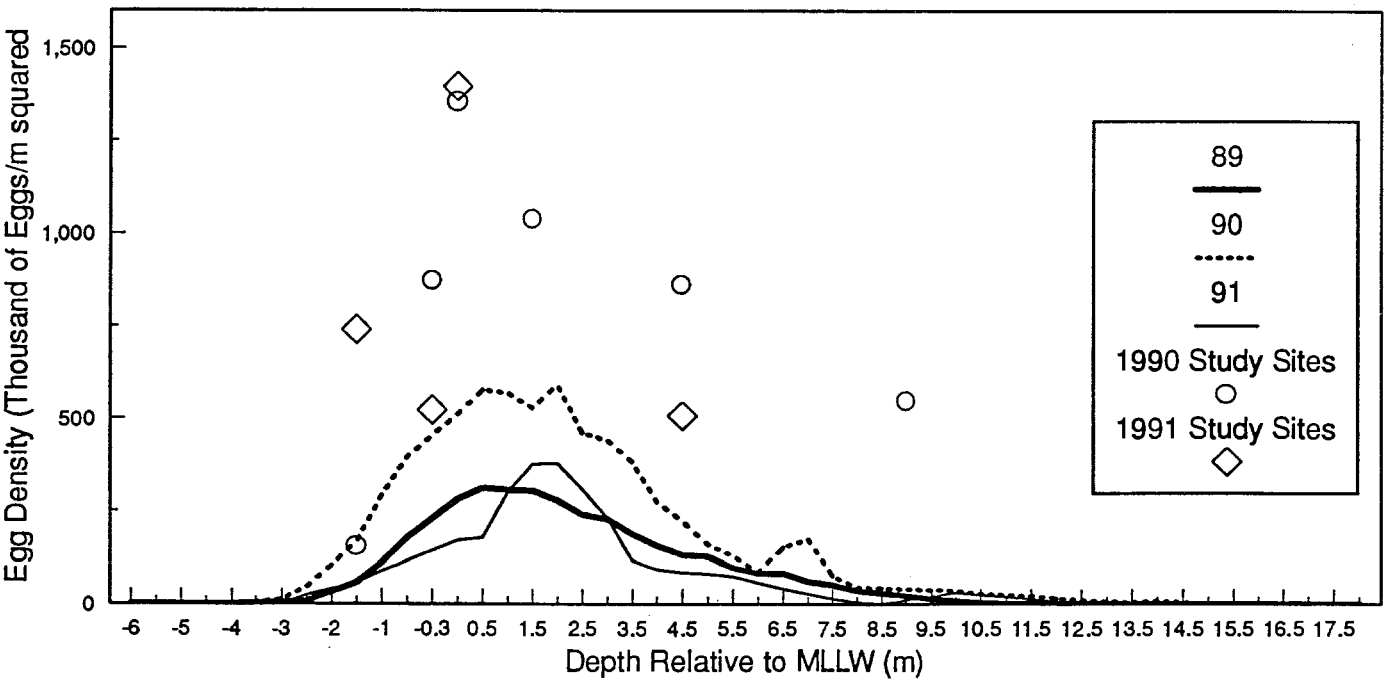


Fig. 2

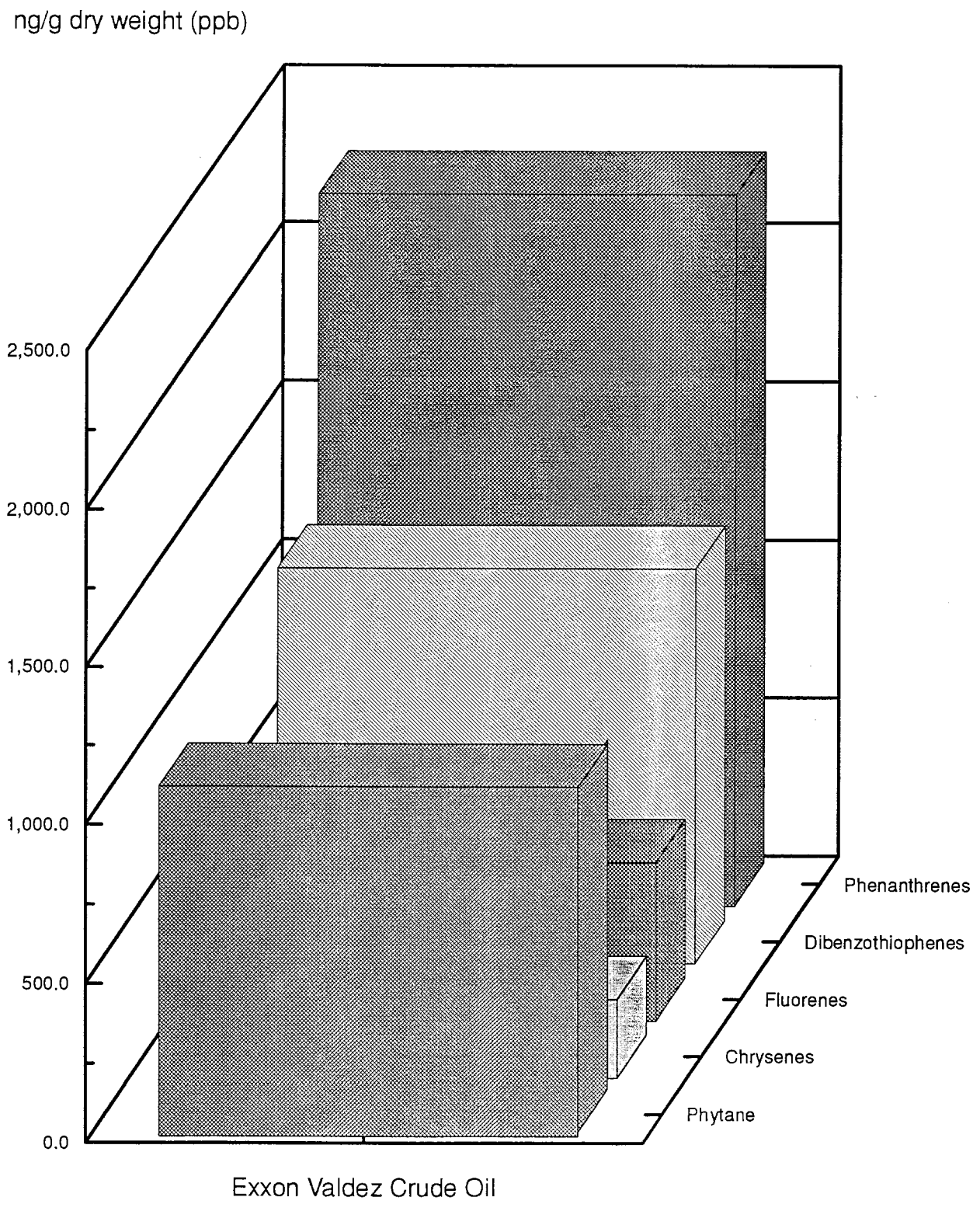


Figure 4. Mean levels of aromatic hydrocarbon groups and phytane for *Exxon Valdez* crude oil (Modified from Karinen et al. 1993).

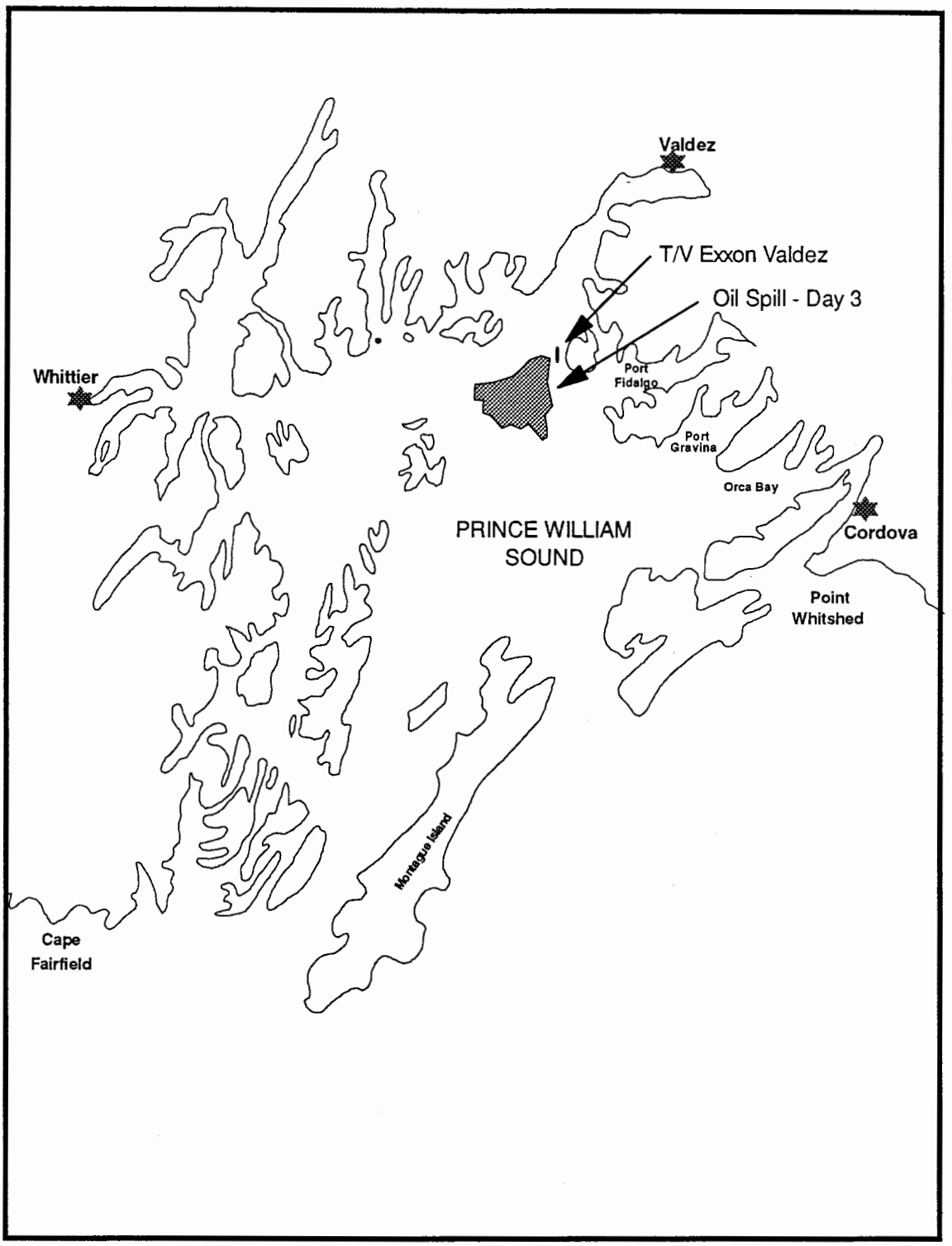


Figure 5. The oil trajectory on March 26, Day 3, of the *Exxon Valdez* spill. (NOAA overflight; Galt/Dahlin/Payton,1300-1500 hrs, Exxon overflight, Teal, 1800 hrs)

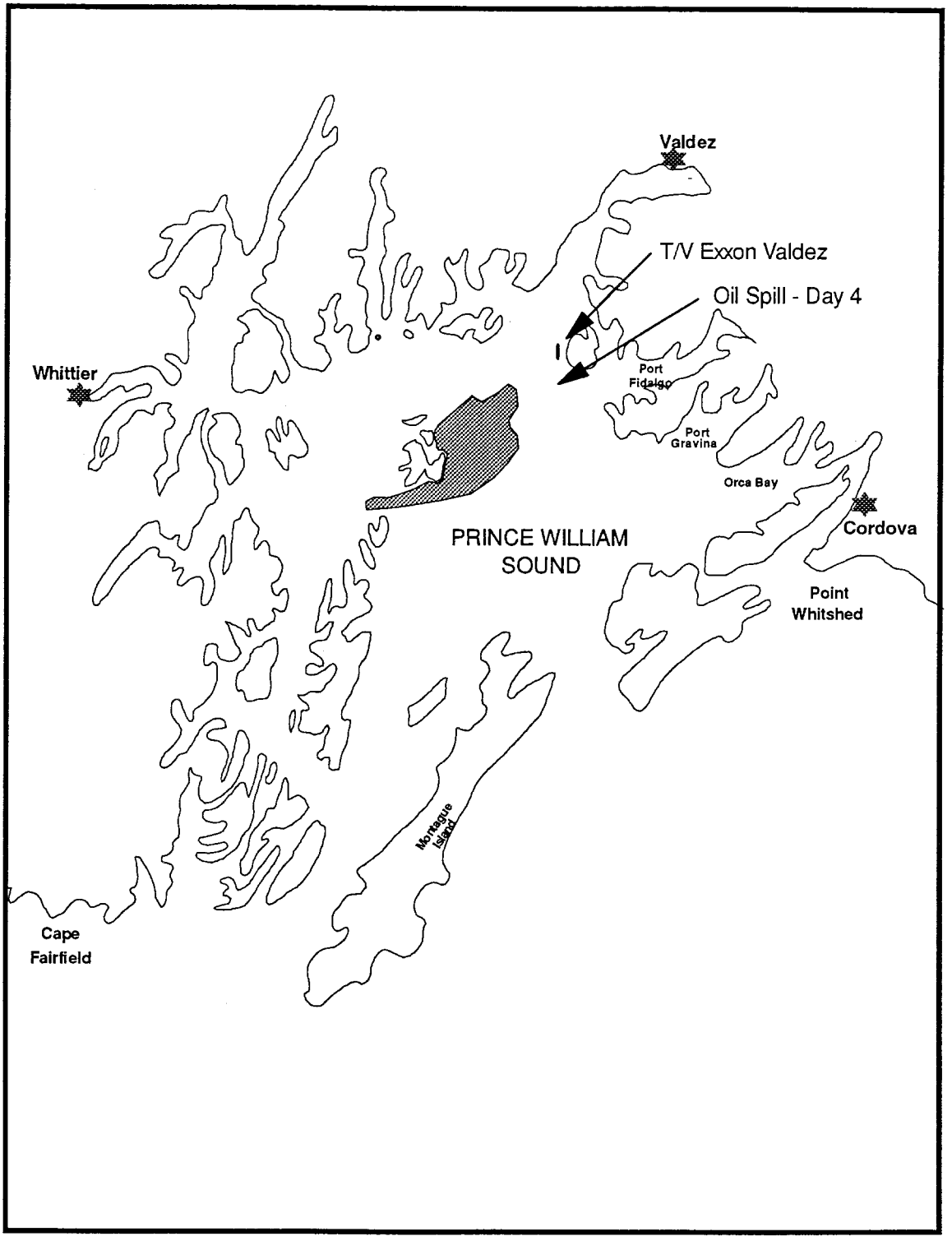


Figure 6. The oil trajectory on March 27, Day 4, of the *Exxon Valdez* spill. (NOAA-DEC Blackhawk overflight Kegler/Sundet/Lockwo/Sotnen/Williams/La, 1130-1315 hrs)

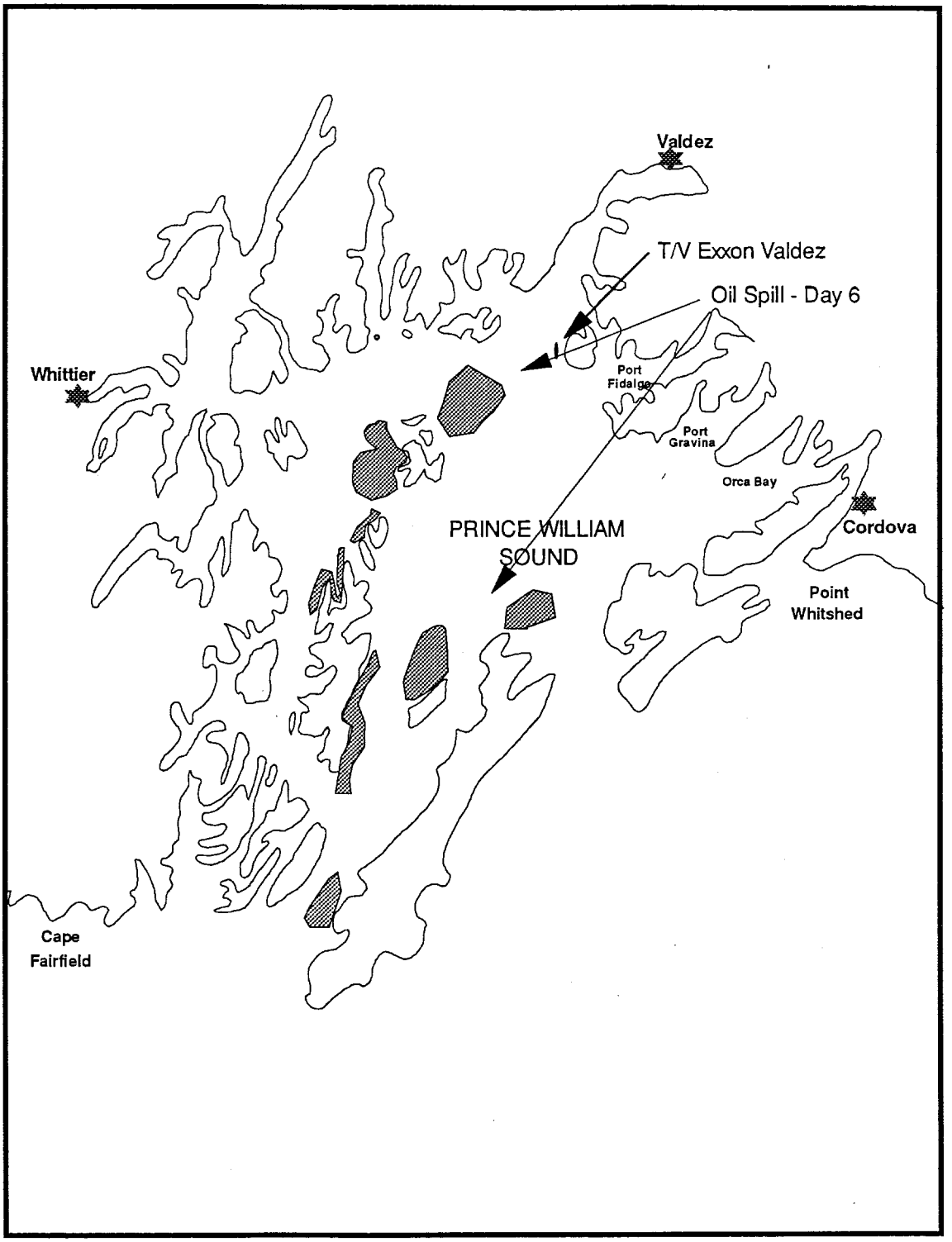


Figure 7. The oil trajectory on March 29, Day 6, from the *Exxon Valdez* spill. (NOAA Twin Otter overflight, Galt & Parker, 1400-1630 hrs)

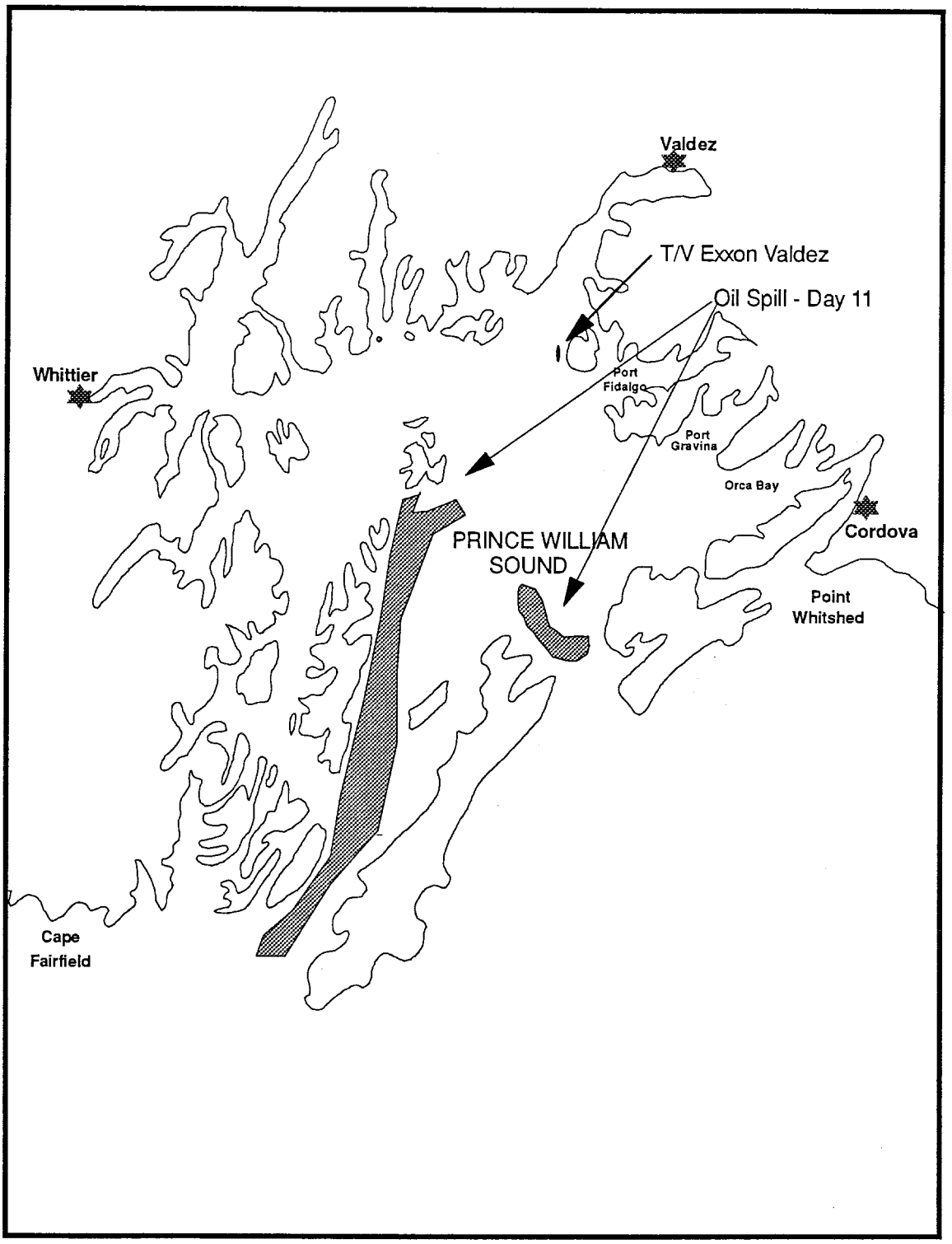


Figure 8. The oil trajectory on April 4, Day 11, from the *Exxon Valdez* spill. (NOAA Overflight, Coast Guard Helicopter 3, Galt & Callahan 1100 - 1300 hrs.)

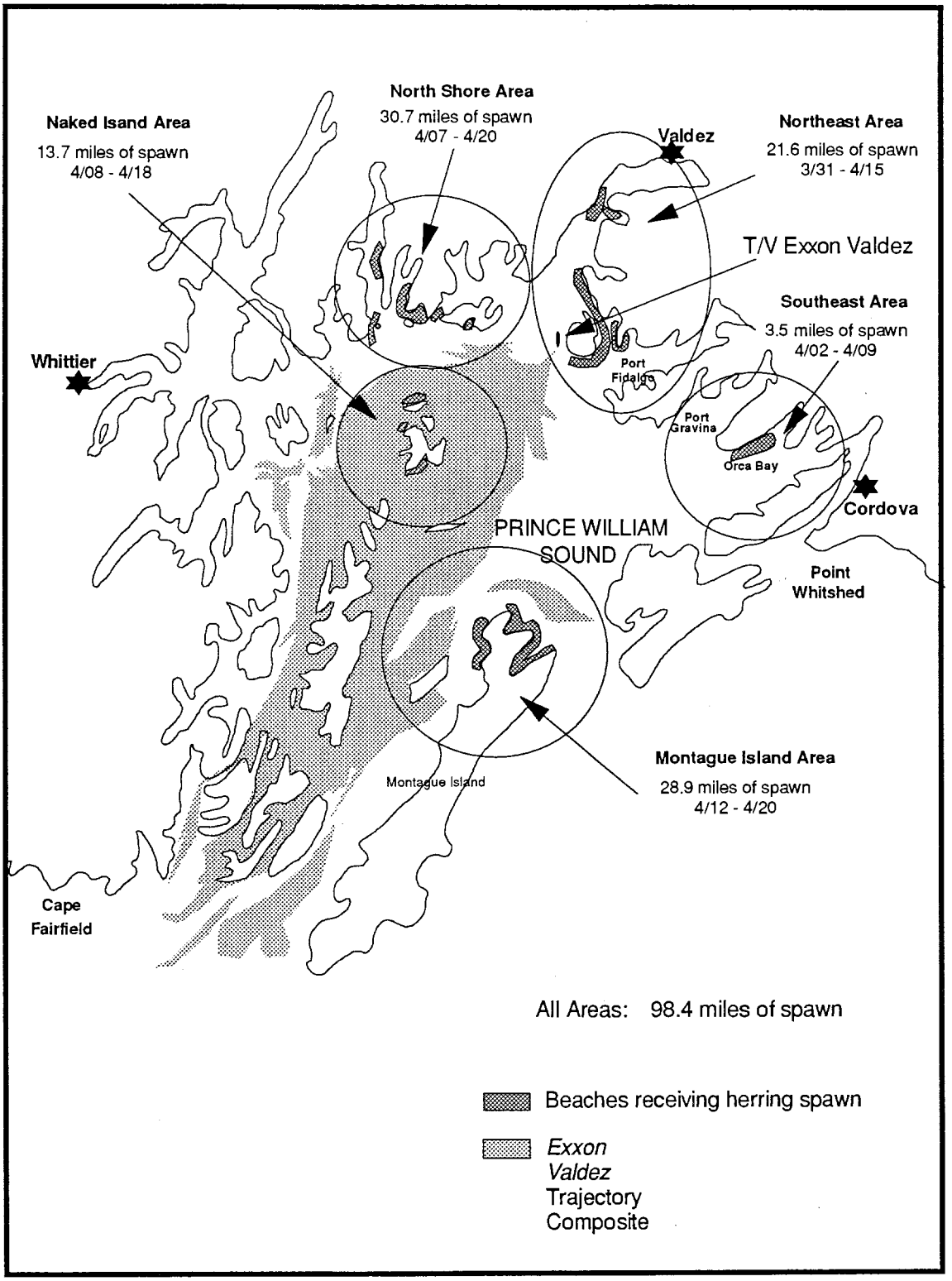


Figure 9. The *Exxon Valdez* oil spill trajectory composite and Prince William Sound herring spawn in 1989.

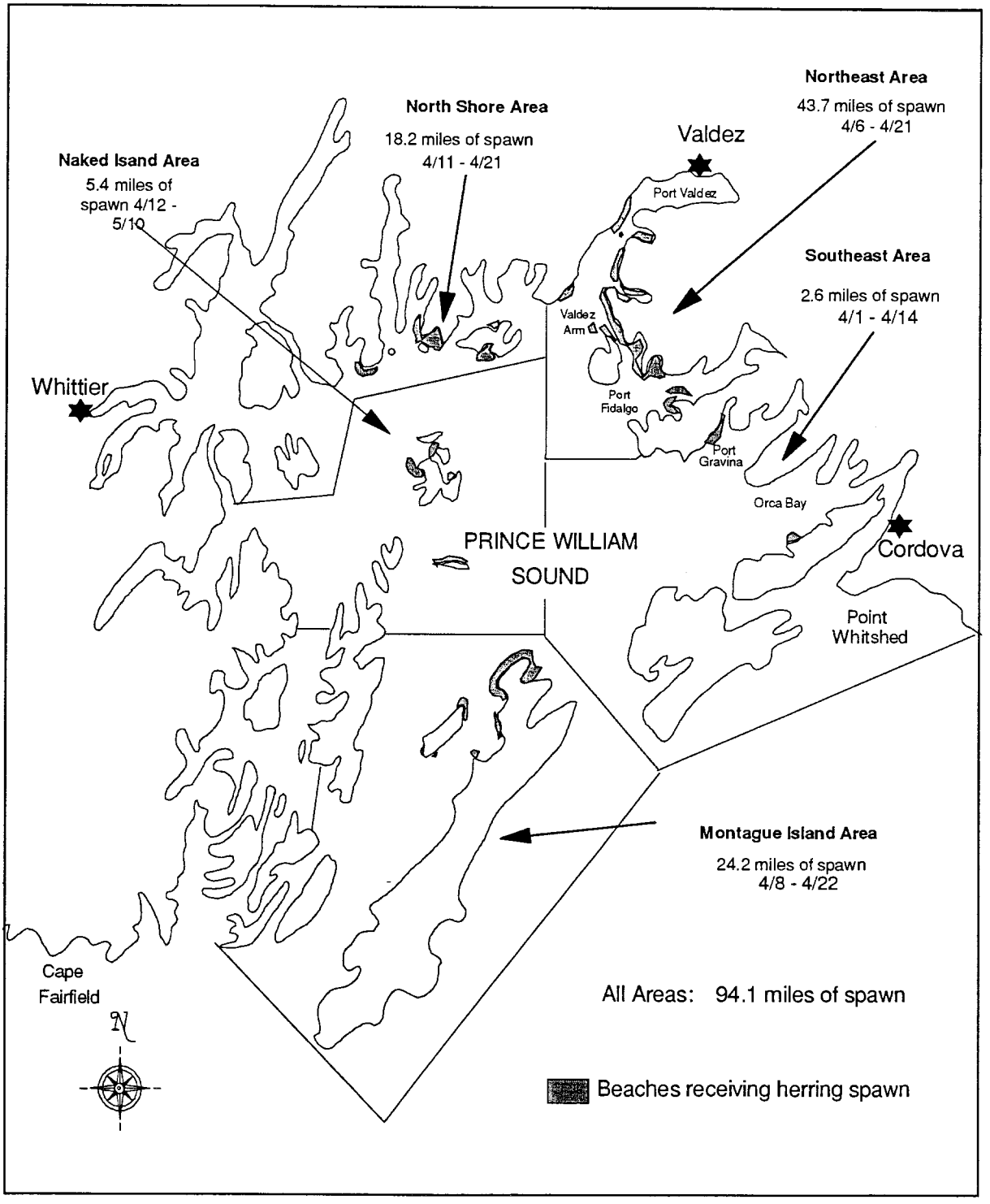


Figure 10. Prince William Sound herring spawn and shoreline mileage mapped by aerial and skiff surveys and the delineation of the five spawning areas used in the calculation of the spawn deposition biomass estimates, 1990.

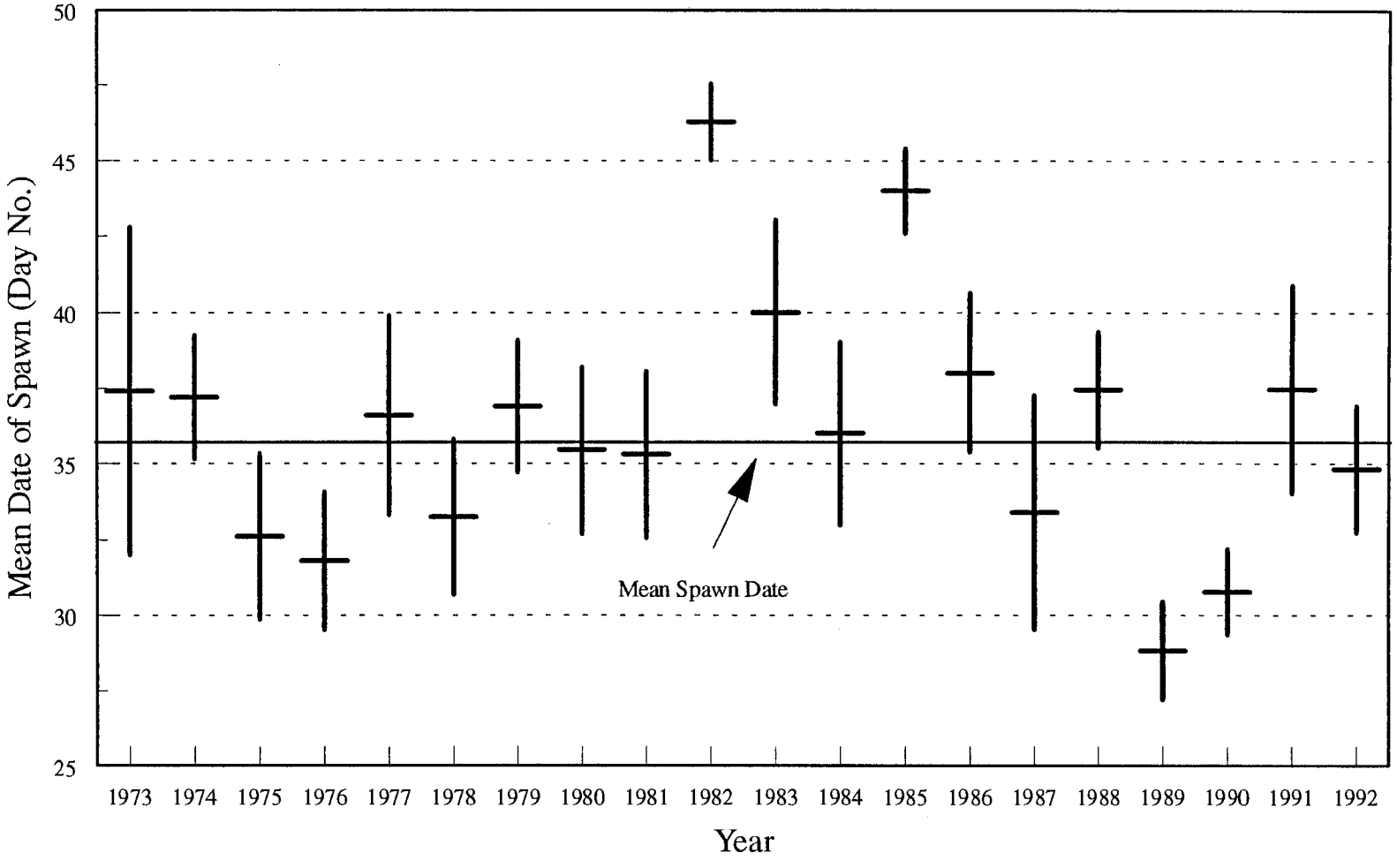


Figure 11. Mean date of herring spawn in Prince William Sound and estimates of error for a 95% C.I., 1973 - 1992.

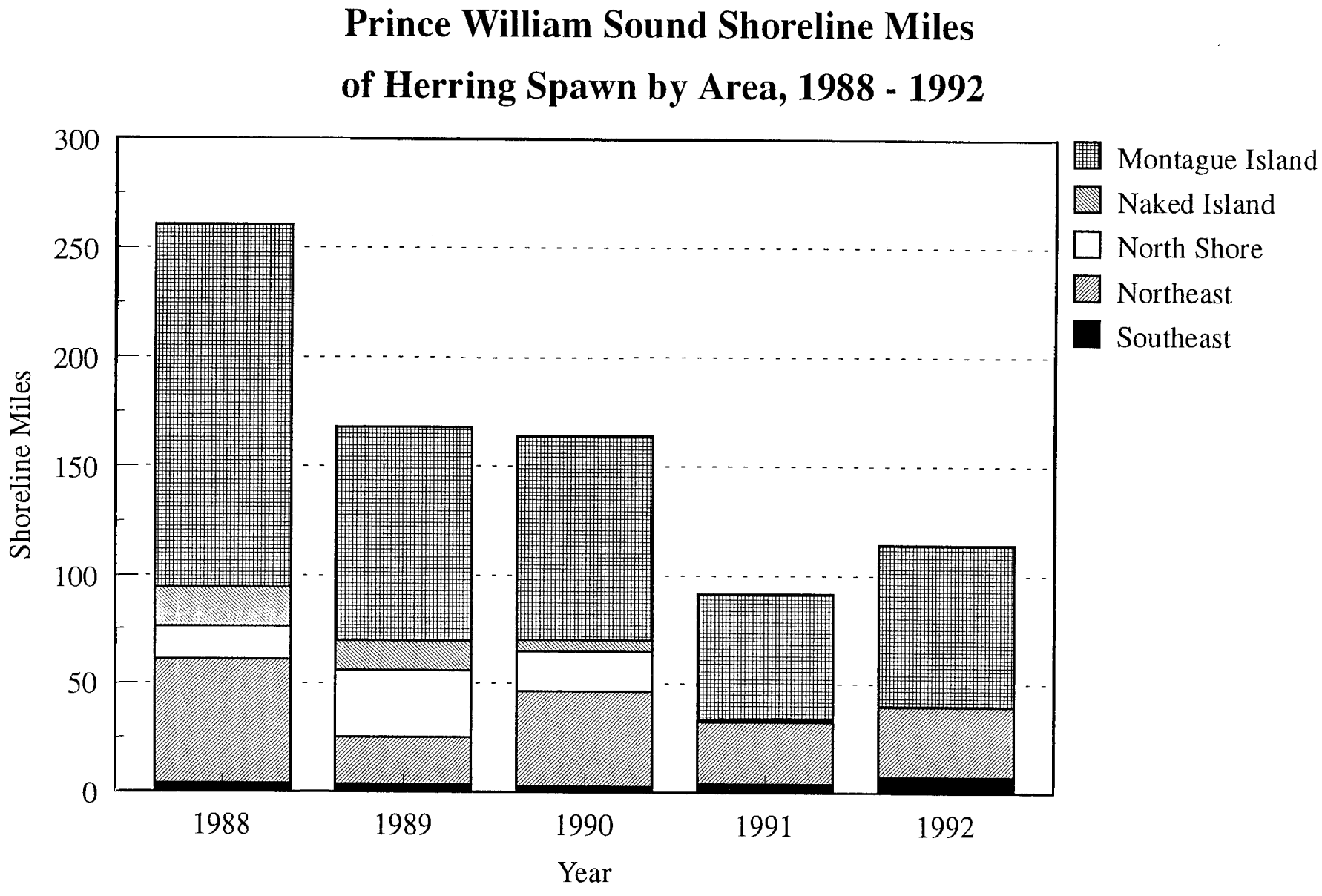


Figure 12. Shoreline miles of herring spawn by area in Prince William Sound, 1988 - 1992.

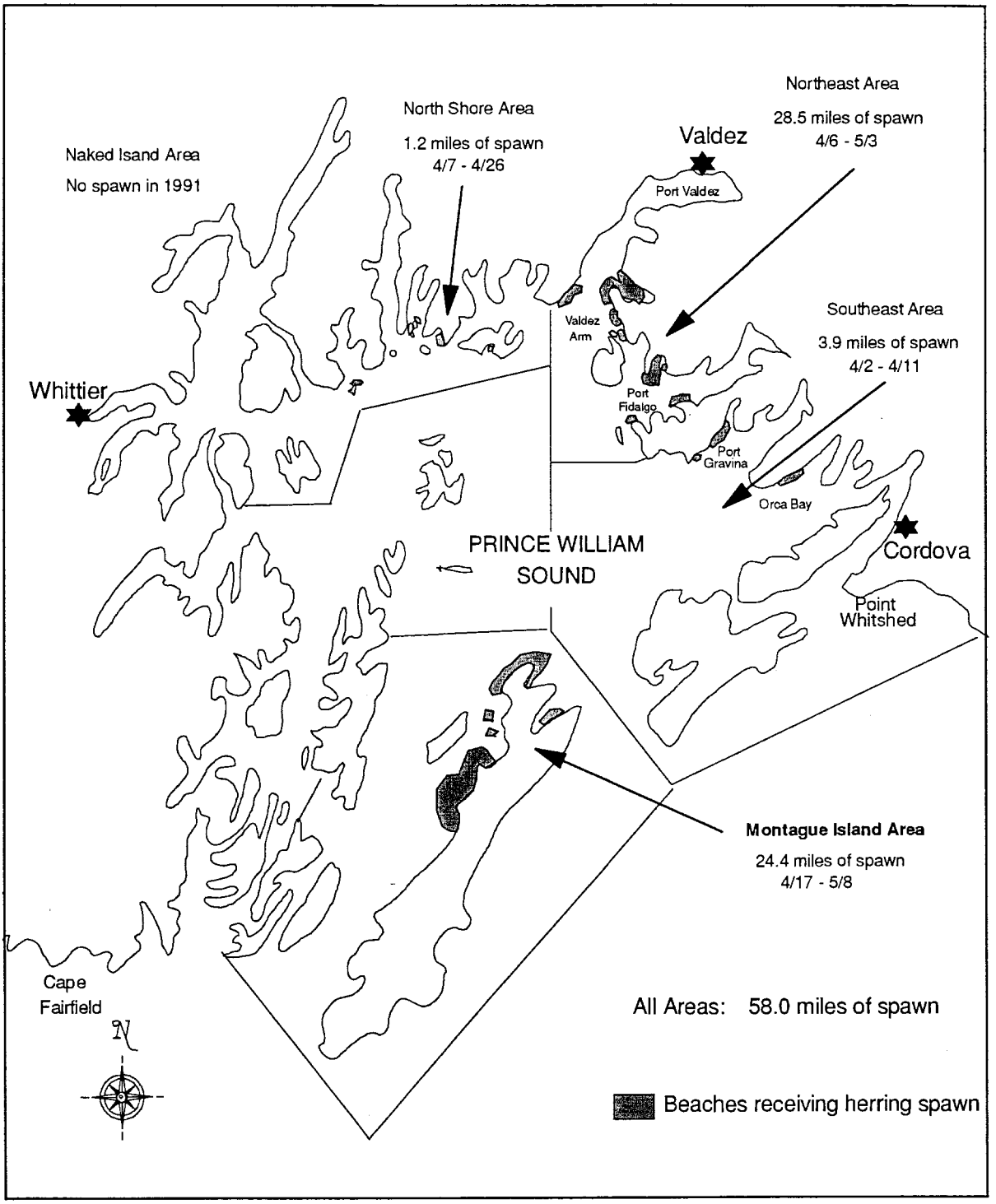


Figure 13. Prince William Sound herring spawn and shoreline mileage mapped by aerial and skiff surveys and the delineation of the five spawning areas used in the calculation of the spawn deposition biomass estimates, 1991.

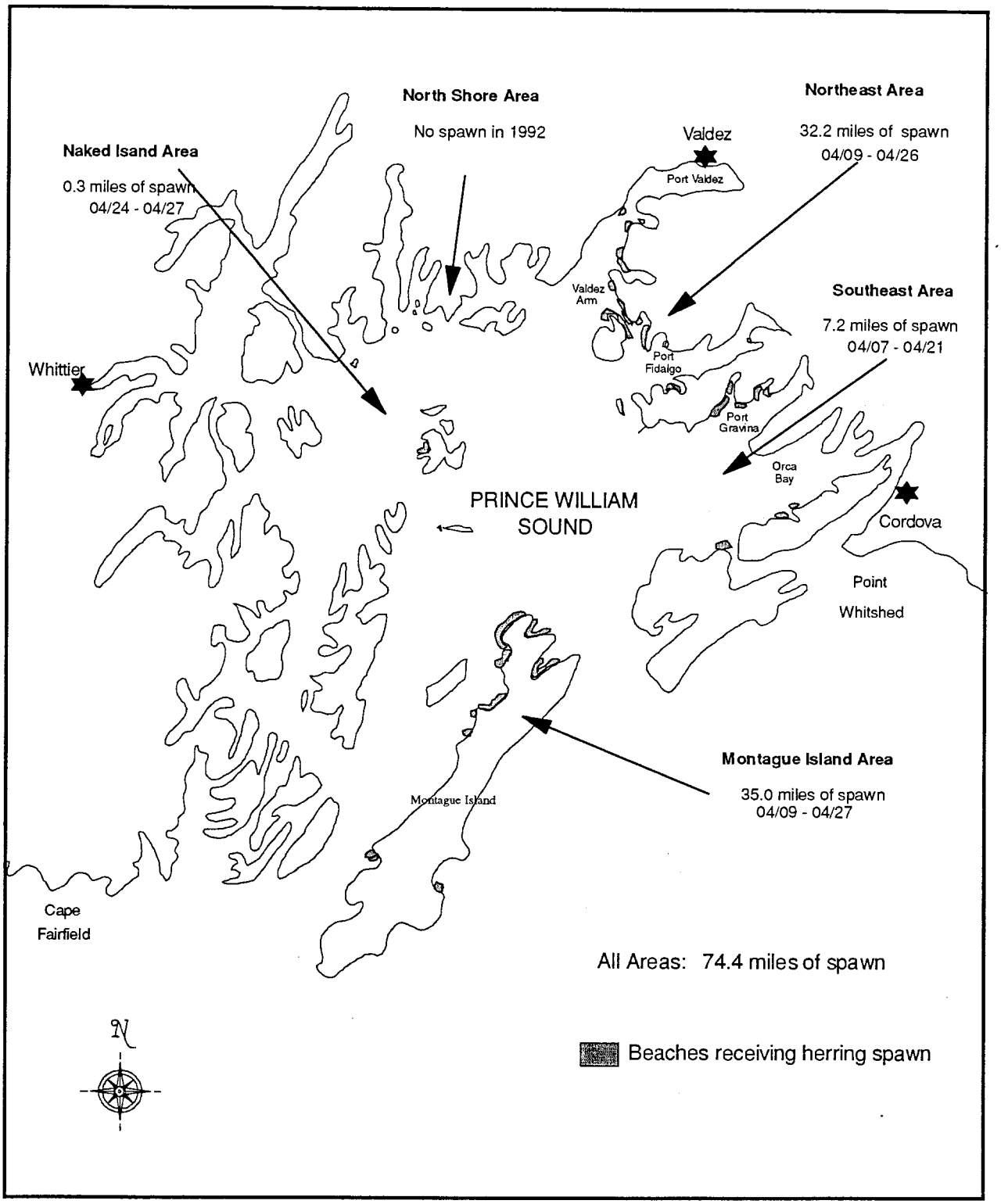


Figure 14. Prince William Sound herring spawn and shoreline mileage mapped by aerial surveys and the delineation of the five spawning areas used in the calculation of the spawn deposition biomass estimates, 1992.

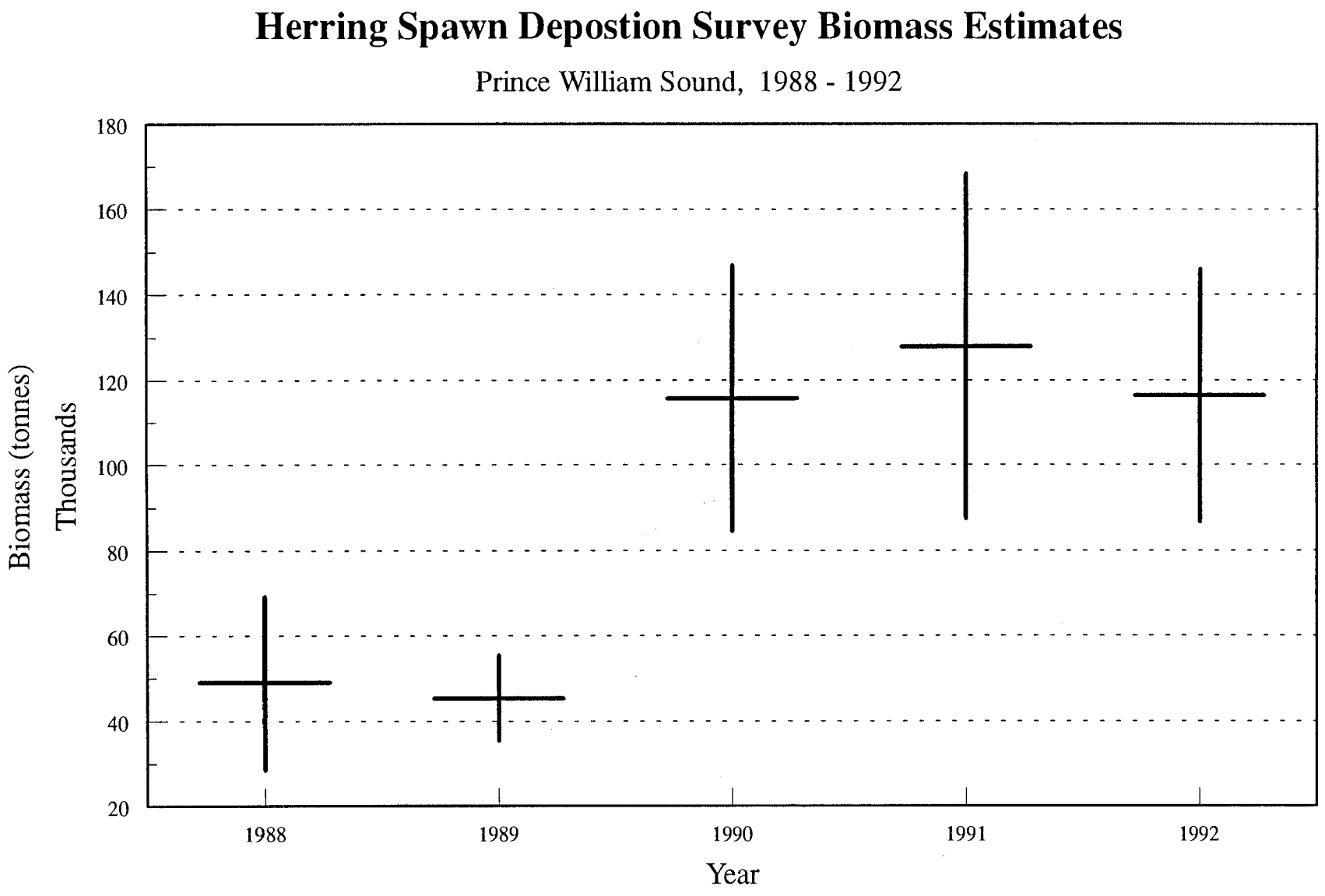
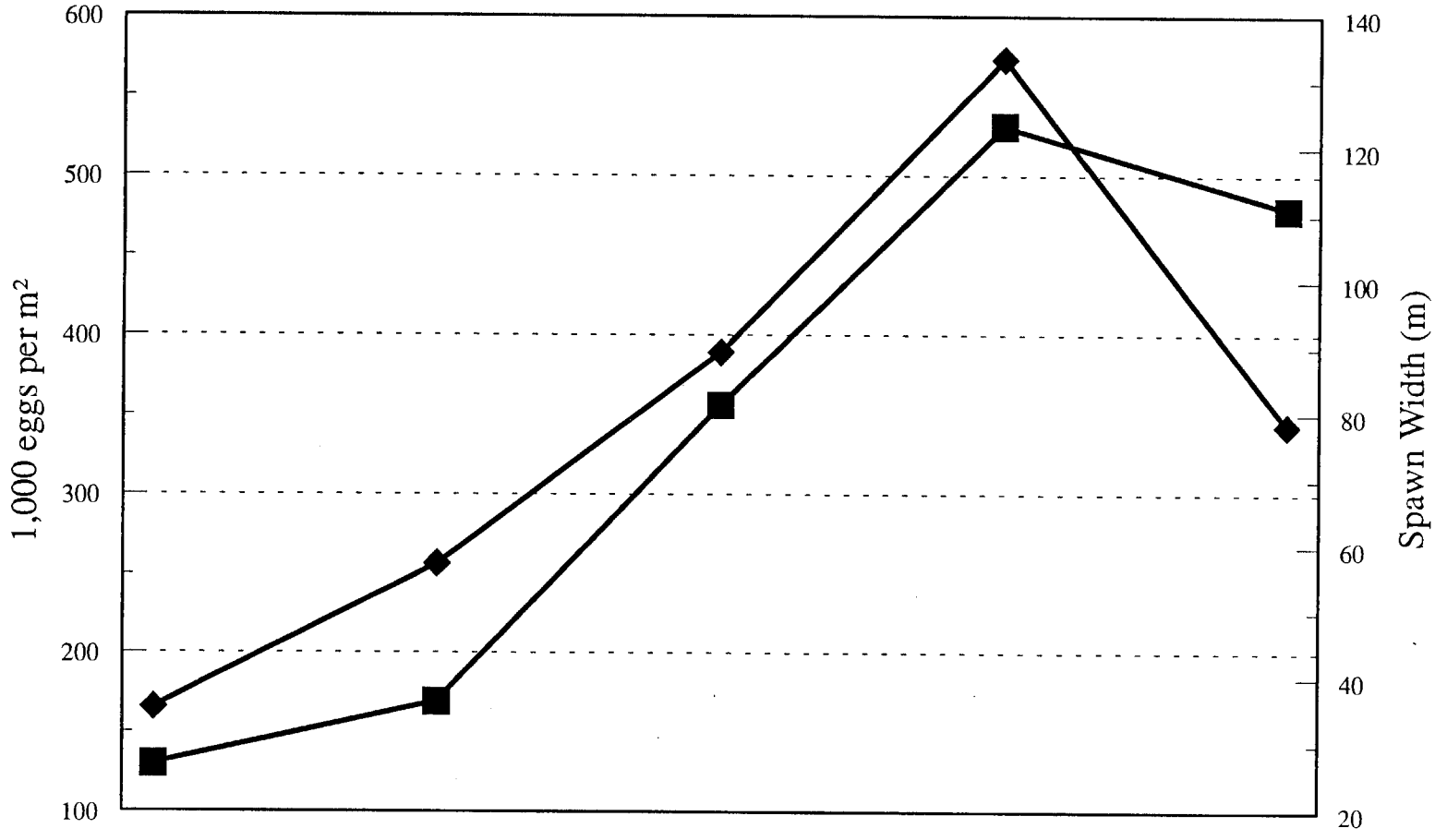


Figure 15. The herring spawn deposition survey escapement biomass estimate in Prince William Sound, 1988 to 1992, with the lower and upper limits of the 95% confidence interval.

Density of Herring Eggs in Spawning Areas

Prince William Sound, 1988-1992



Year	1988	1989	1990	1991	1992
Egg Density (1000/m²) ■	130.0	169.0	356.0	531.0	479.0
Width of Spawn (m) ◆	35.7	57.5	89.4	133.5	78.3

Figure 16. Average density of herring eggs in spawning areas within Prince William Sound, 1988 - 1992

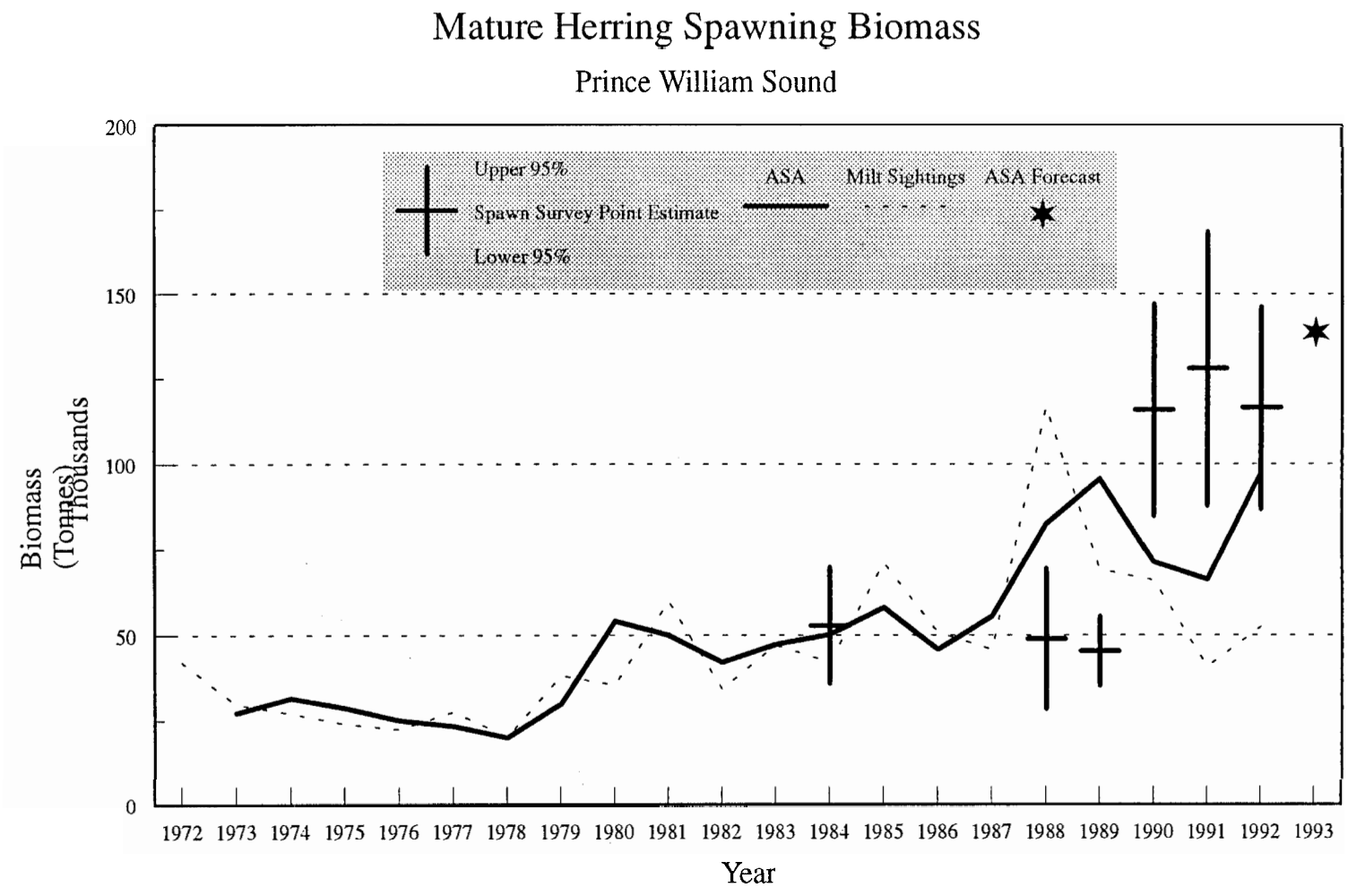


Figure 17. Mature biomass estimated by the ASA model, by spawn deposition surveys, and by aerial milt surveys, after scaling by the milt survey coefficient estimated by the ASA model, 1972-1993 (Fritz Funk, Alaska Department of Fish and Game, personal communication).

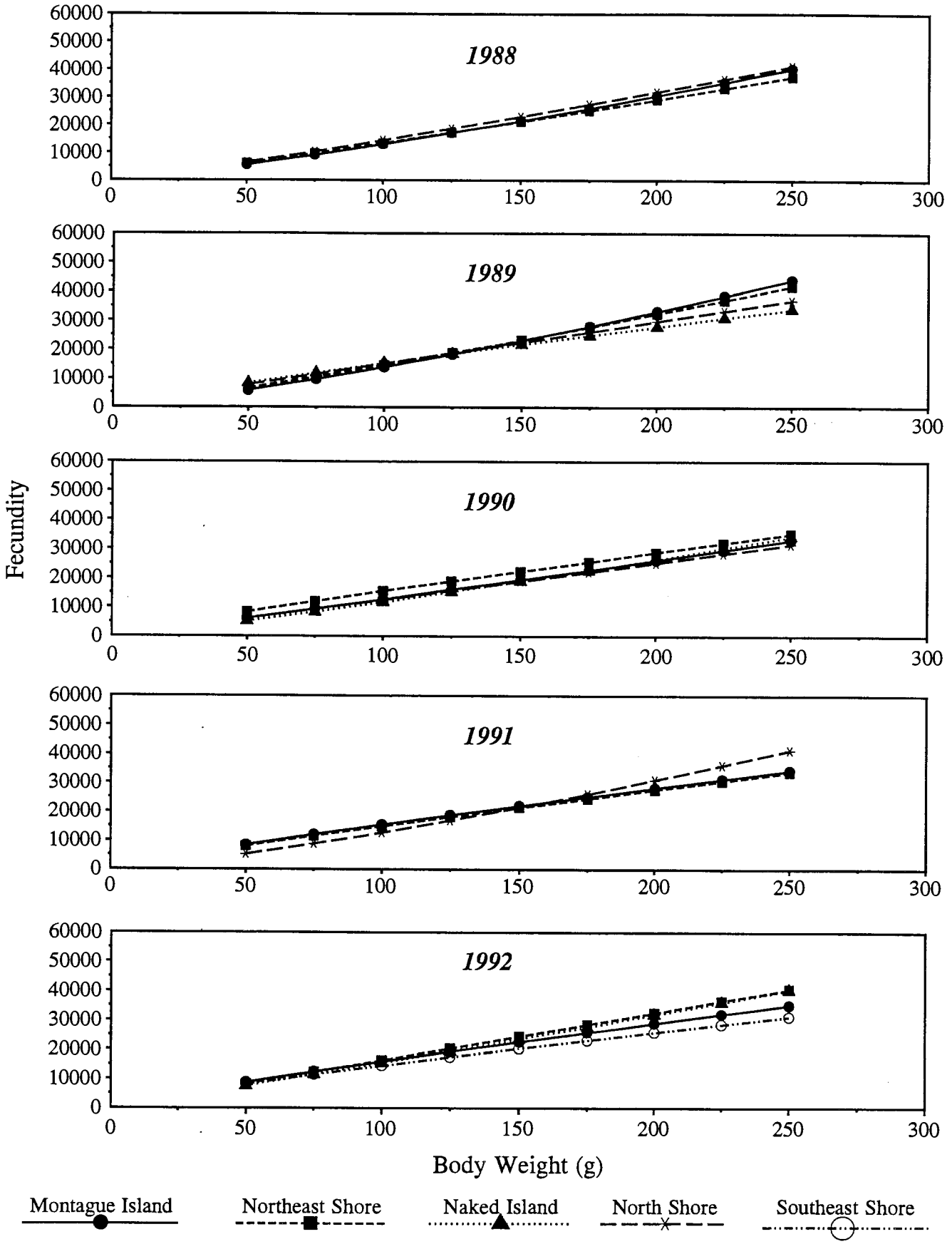


Figure 18. Relationship between fecundity and body weight of female Pacific herring, by year, in Prince William Sound, Alaska.

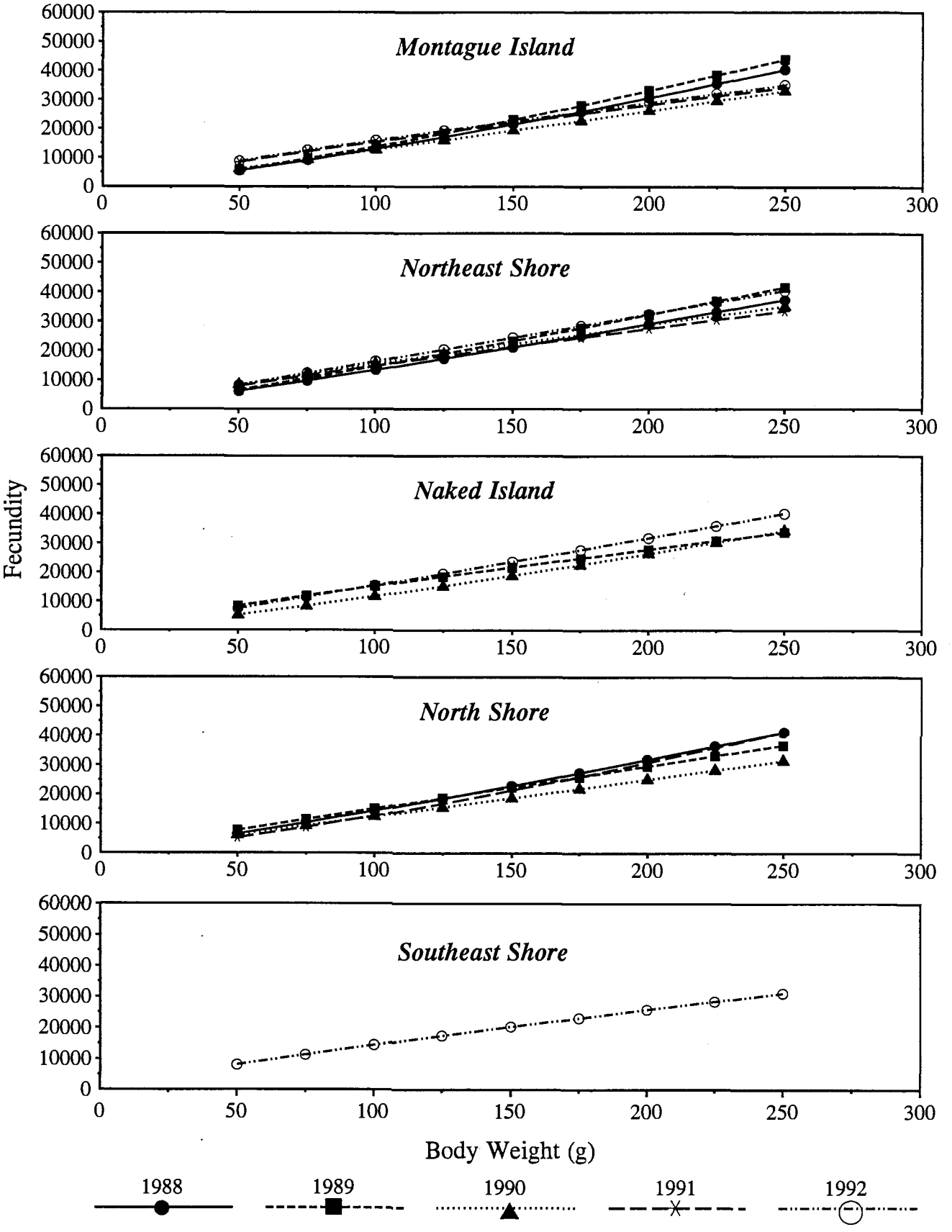


Figure 19. Relationship between fecundity and body weight of female Pacific herring, by area, in Prince William Sound, Alaska.

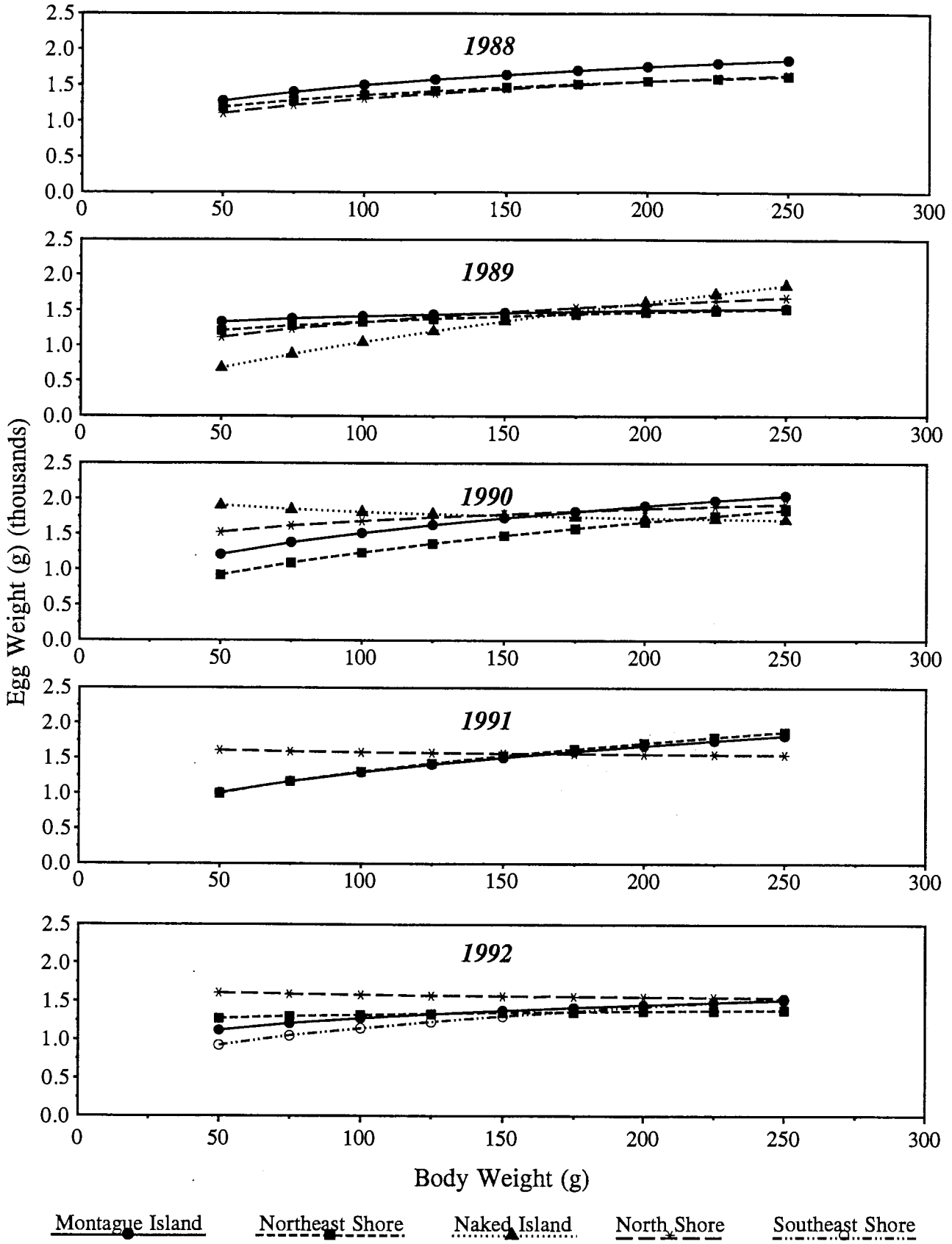


Figure 20. Relationship between egg and body weight of female Pacific herring, by year, in Prince William Sound, Alaska.

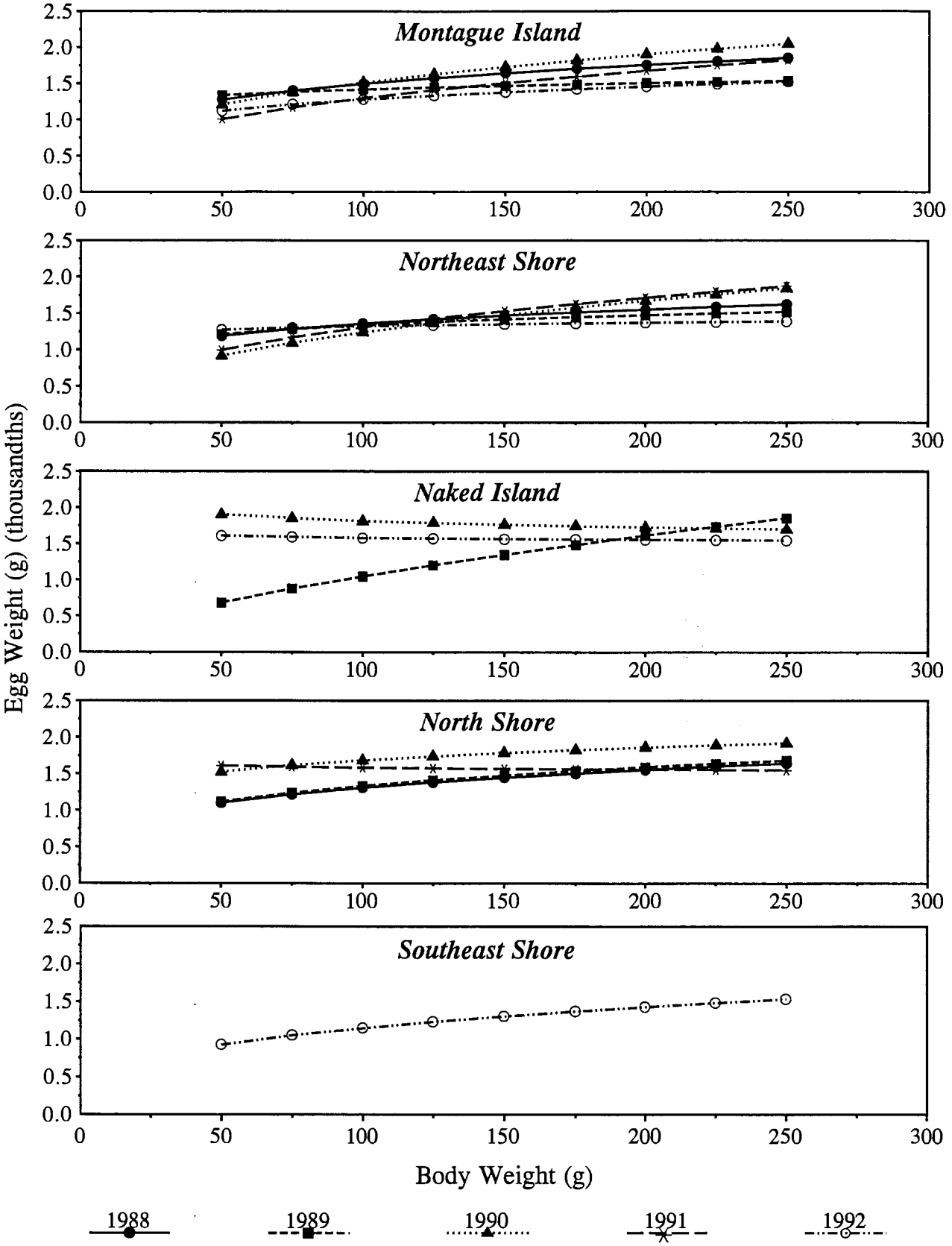


Figure 21. Relationship between egg and body weight of female Pacific herring, by area, in Prince William Sound, Alaska.

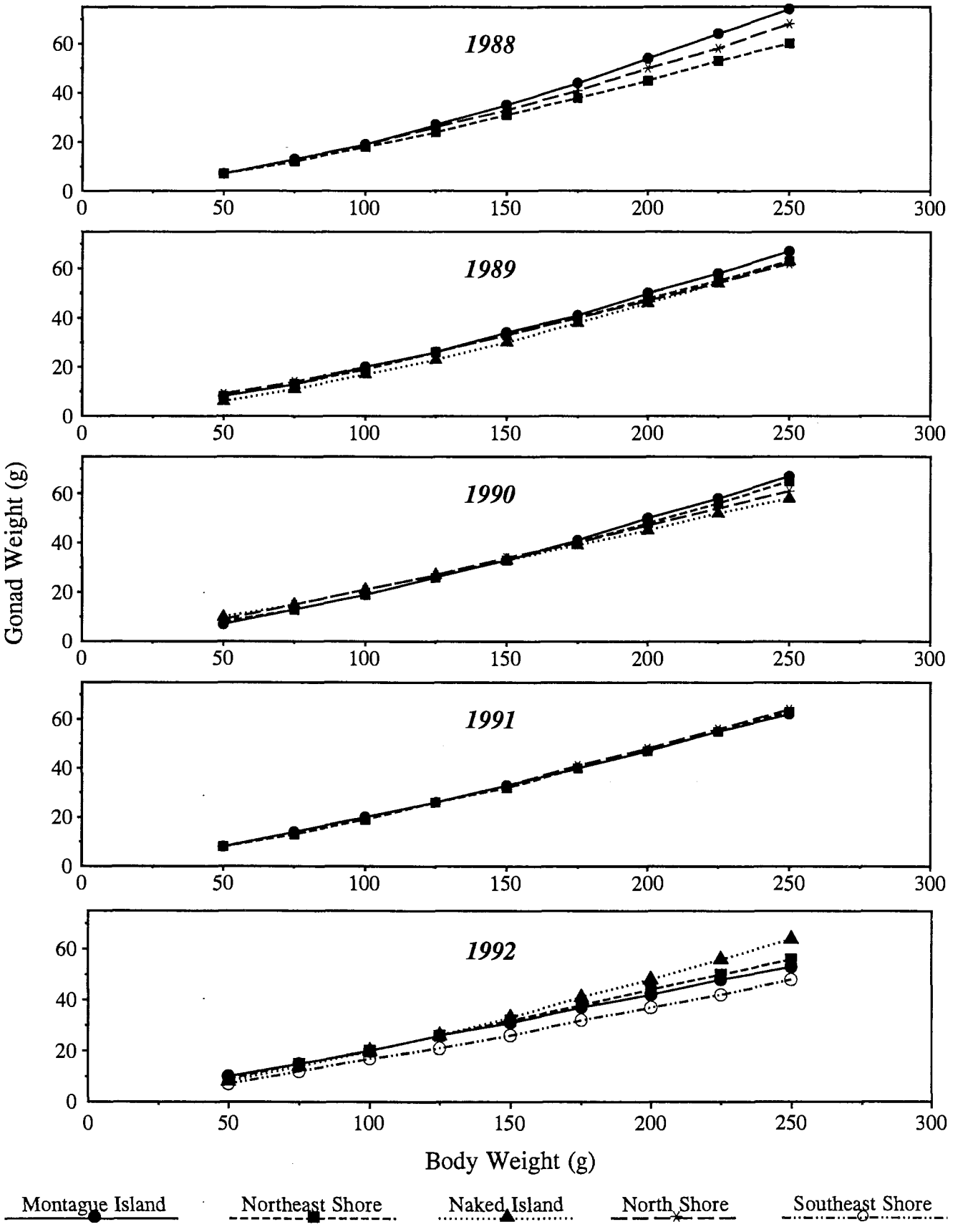


Figure 22. Relationship between gonad and body weight of female Pacific herring, by year, in Prince William Sound, Alaska.

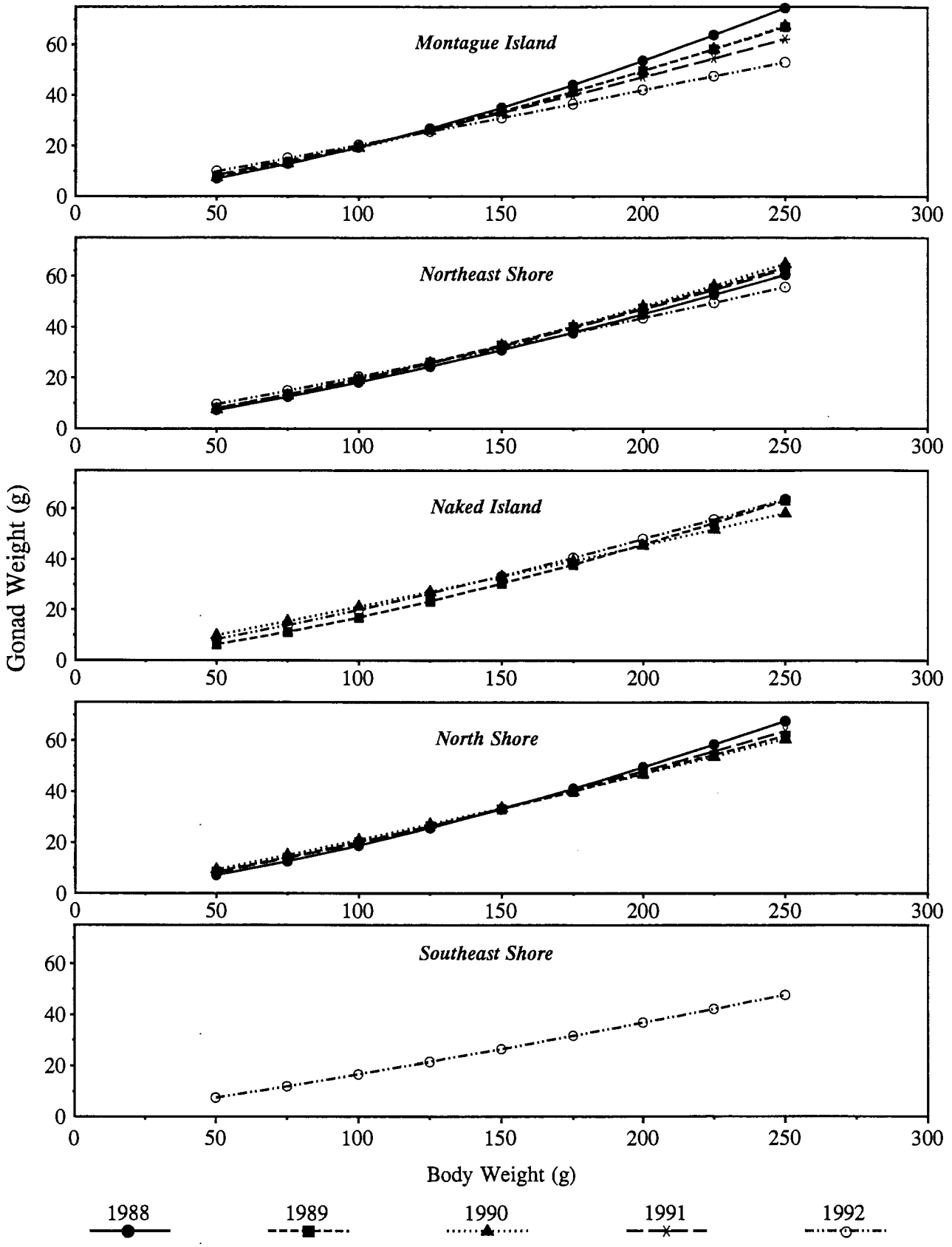


Figure 23. Relationship between gonad and body weight of female Pacific herring, by area, in Prince William Sound, Alaska.

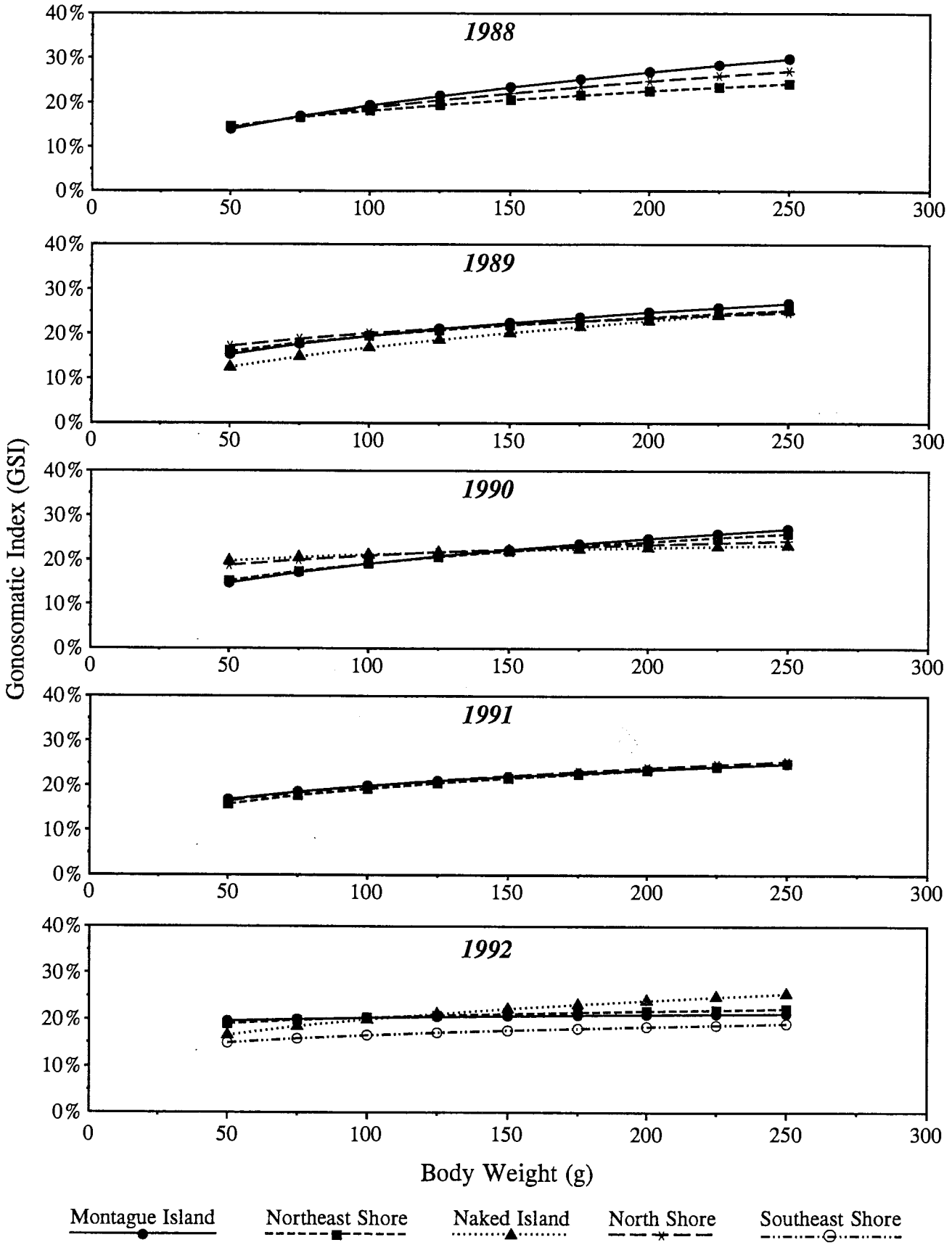


Figure 24. Relationship between gonosomatic index (GSI) and body weight of female Pacific herring, by year, in Prince William Sound, Alaska.

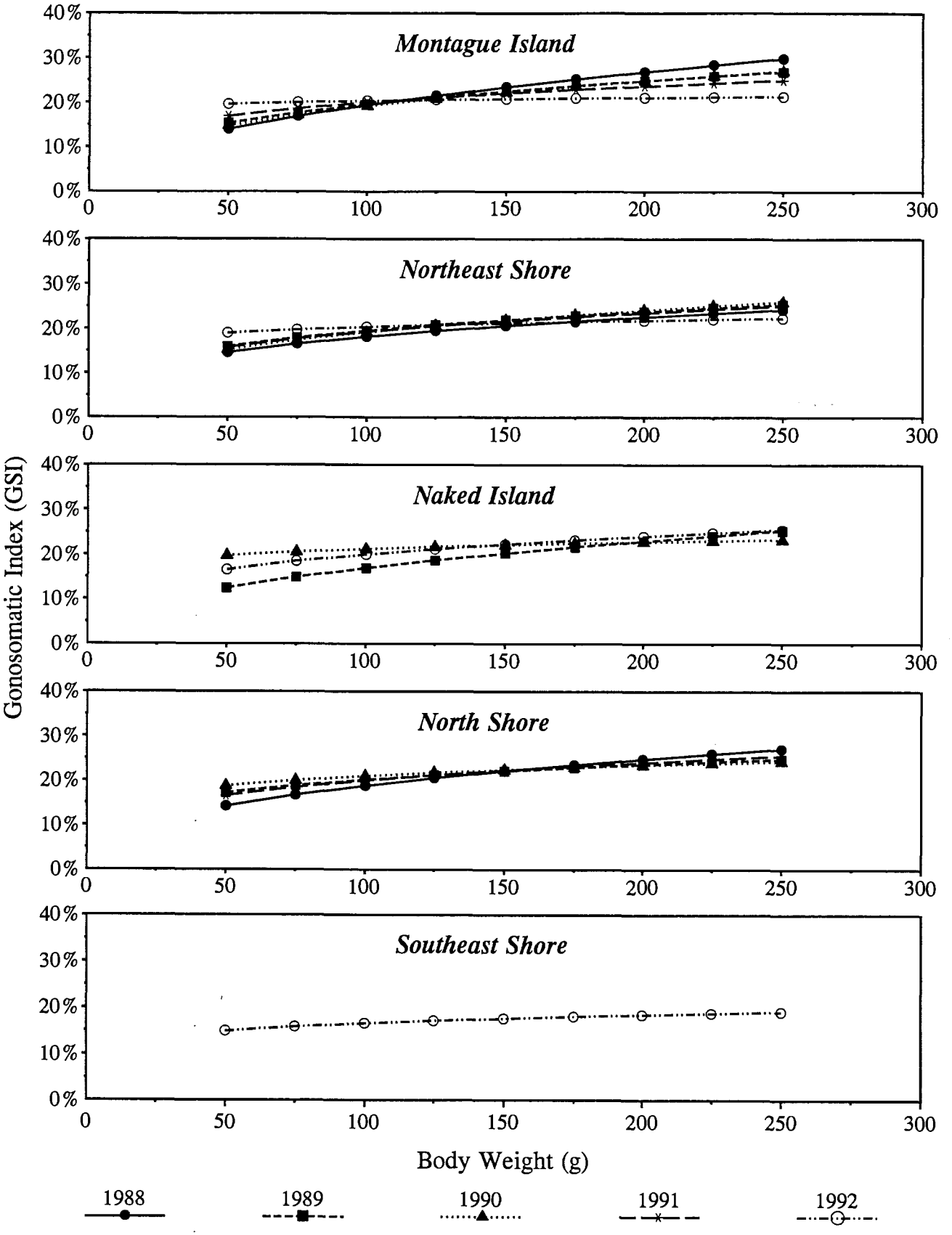


Figure 25. Relationship between gonosomatic index (GSI) and body weight of female Pacific herring, by area, in Prince William Sound, Alaska.

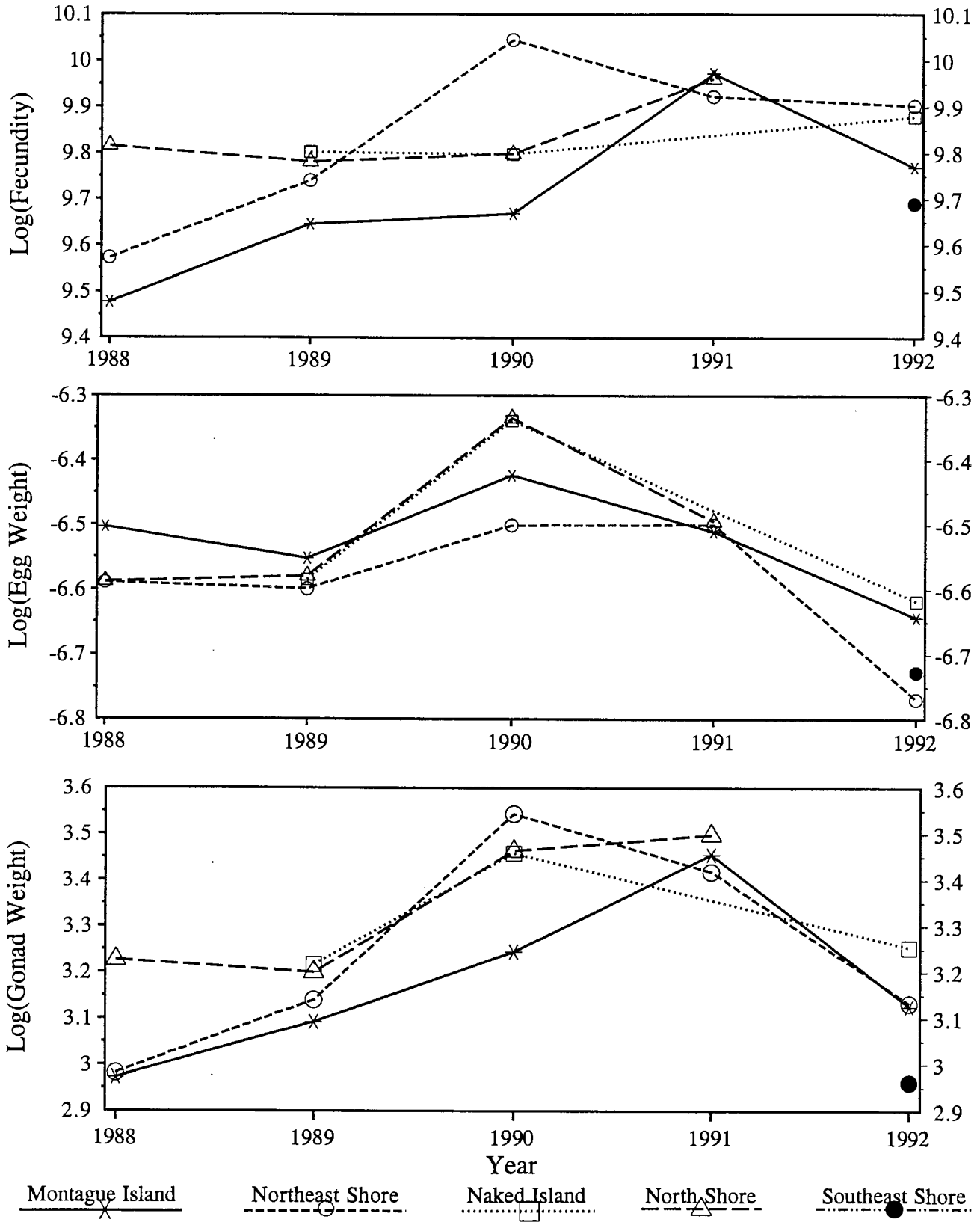


Figure 26. Year and area interactions of least square means for log(fecundity), log(egg weight), and log(gonad weight) for Pacific herring in Prince William Sound, Alaska.

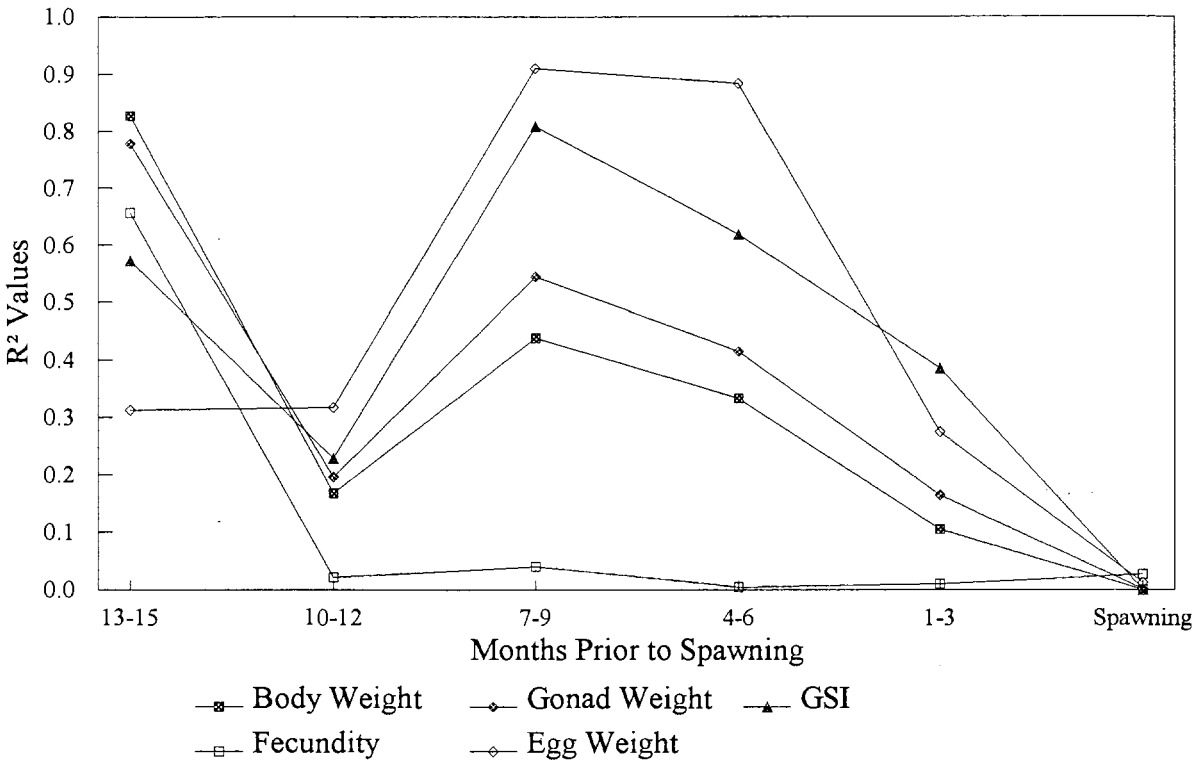


Figure 27. Coefficient of determination (R^2) values for linear regressions using either body weight, gonad weight, gonosomatic Index (GSI), fecundity, or egg weight for Pacific herring in Prince William Sound from 1988-1992 as dependent variables and mean sea surface temperature anomolies (SSTA's) at three grid points in the western Gulf of Alaska (55°N, 160°W; 60°N, 150°W; 60°N, 155°W) as the independent variable. Mean sea surface temperature anomolies were grouped into 3 month periods prior to April, the month during which herring usually spawn within Prince William Sound.

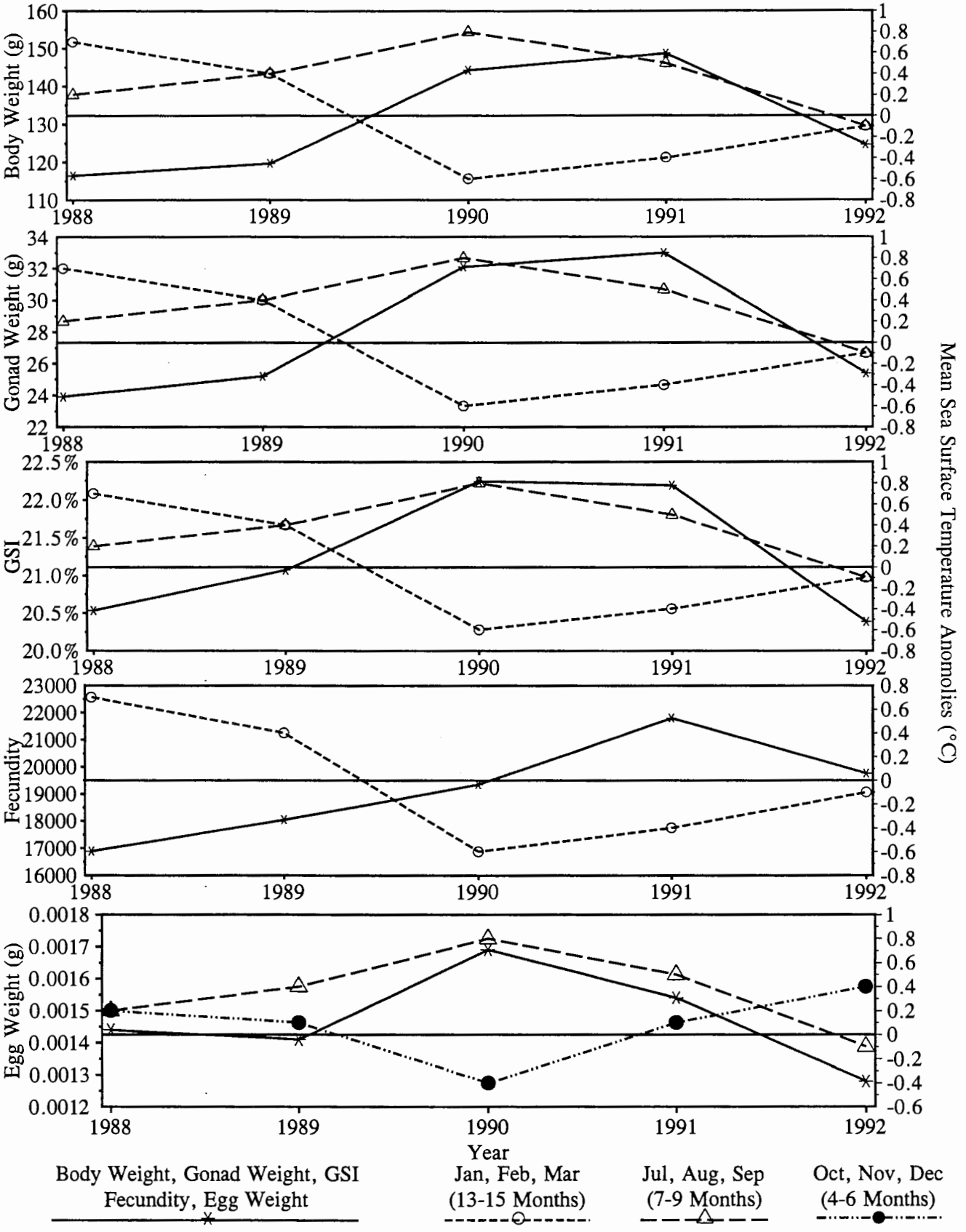


Figure 28. Mean body weight, gonad weight, gonosomatic index (GSI), fecundity, and egg weight plotted against mean sea surface temperature anomolies (°C) at three grid points (55°N, 160°W; 60°N, 150°W; 60°N, 155°W) in the Gulf of Alaska.

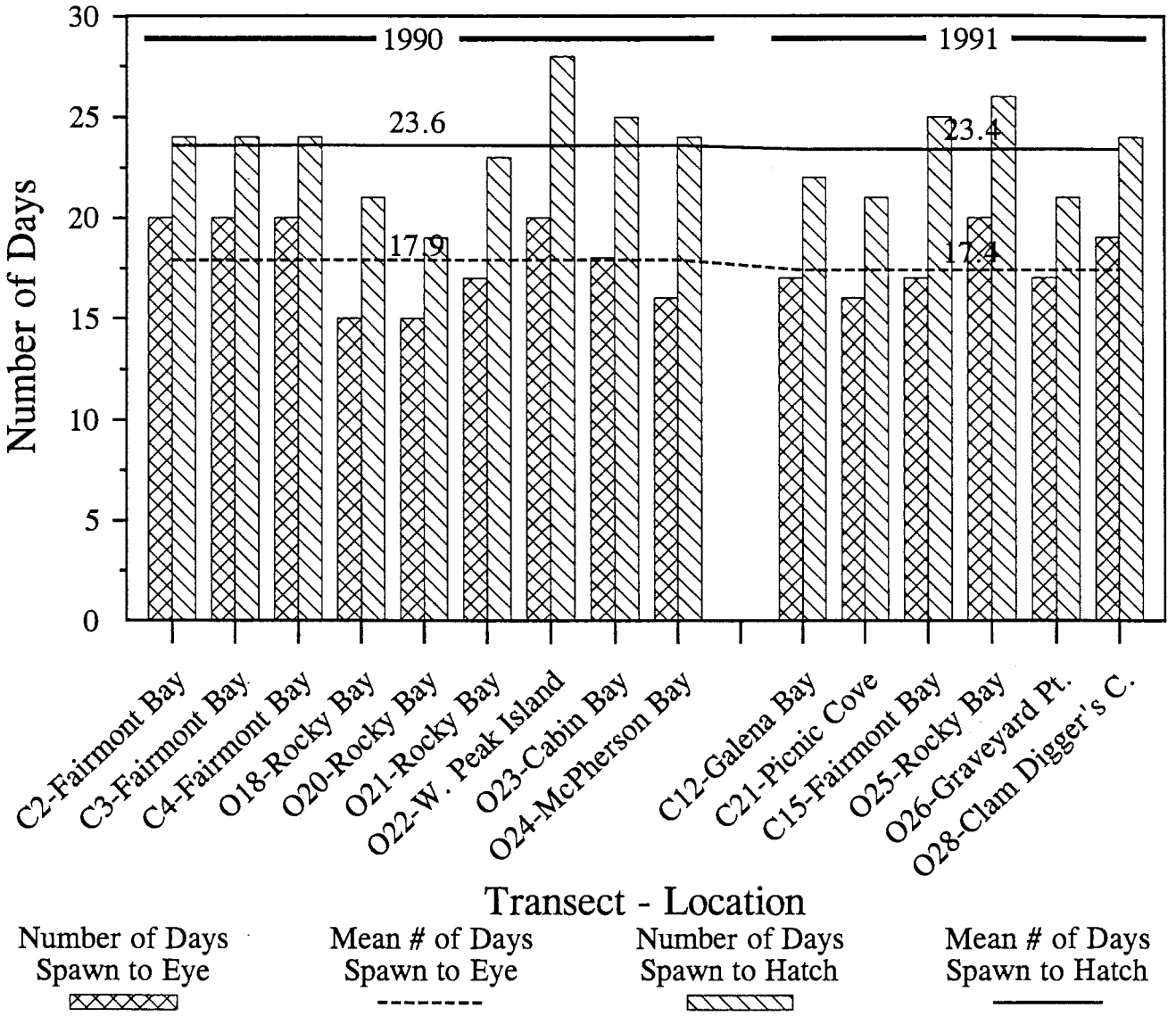


Figure 29. Number of days from first observed spawn to eyed eggs and to eggs hatching at egg loss transect sampling locations in Prince Willian Sound, Alaska, 1990-1991.

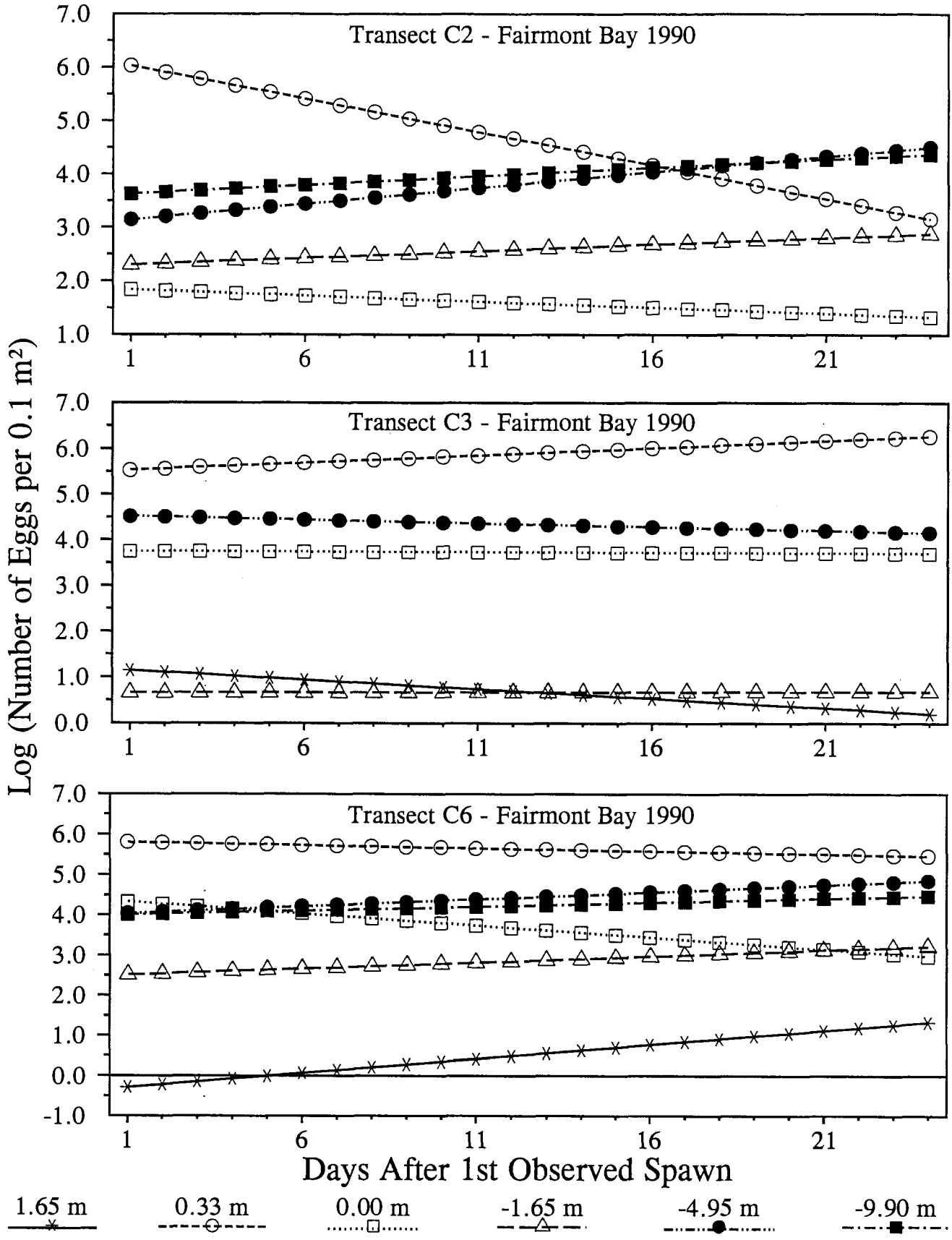


Figure 30. Estimated egg loss, by depth, at transect sampling locations in Fairmont Bay, North Shore area, Prince William Sound, Alaska, 1990.

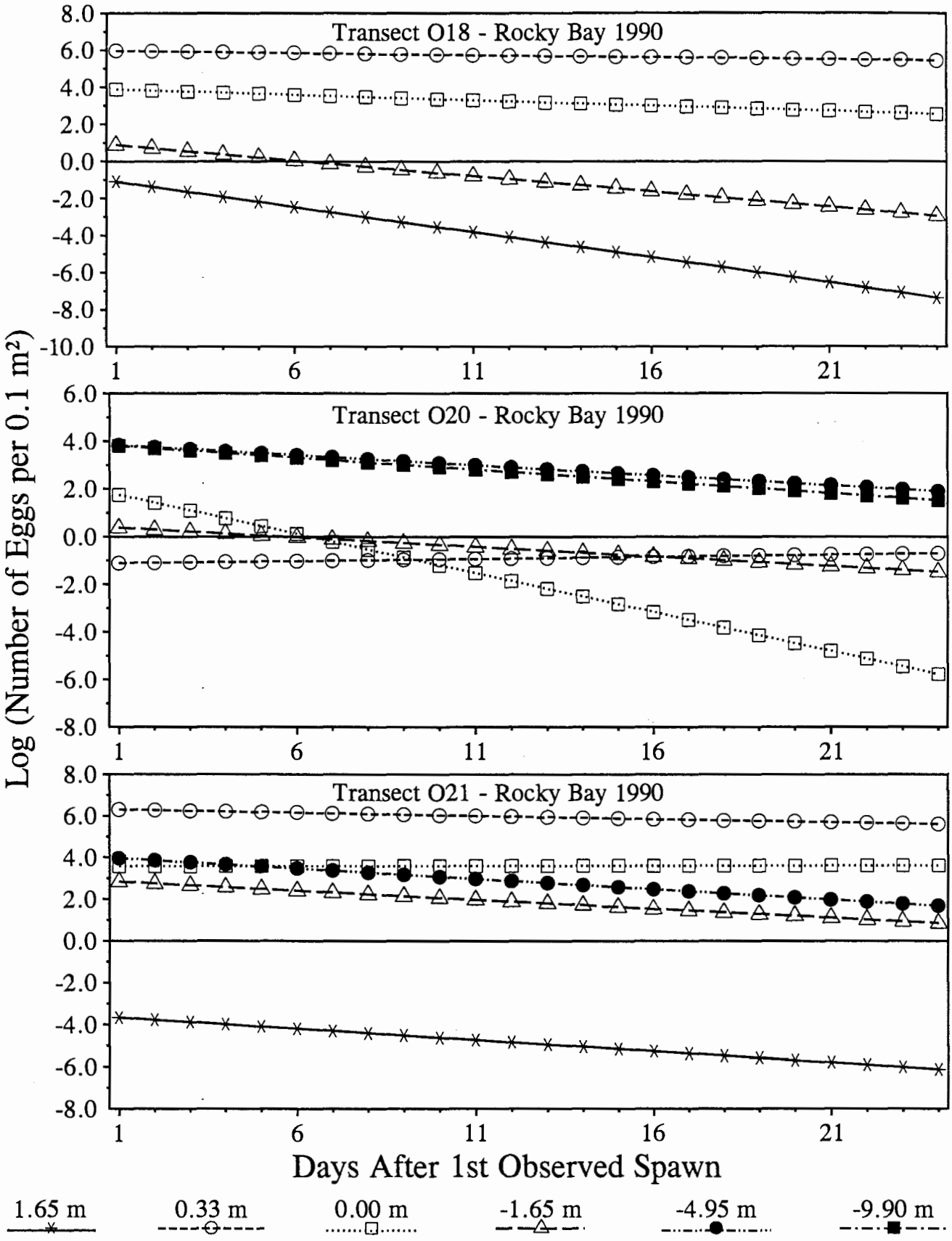


Figure 31. Estimated egg loss, by depth, at transect sampling locations in Rocky Bay, Montague Island area, Prince William Sound, Alaska, 1990.

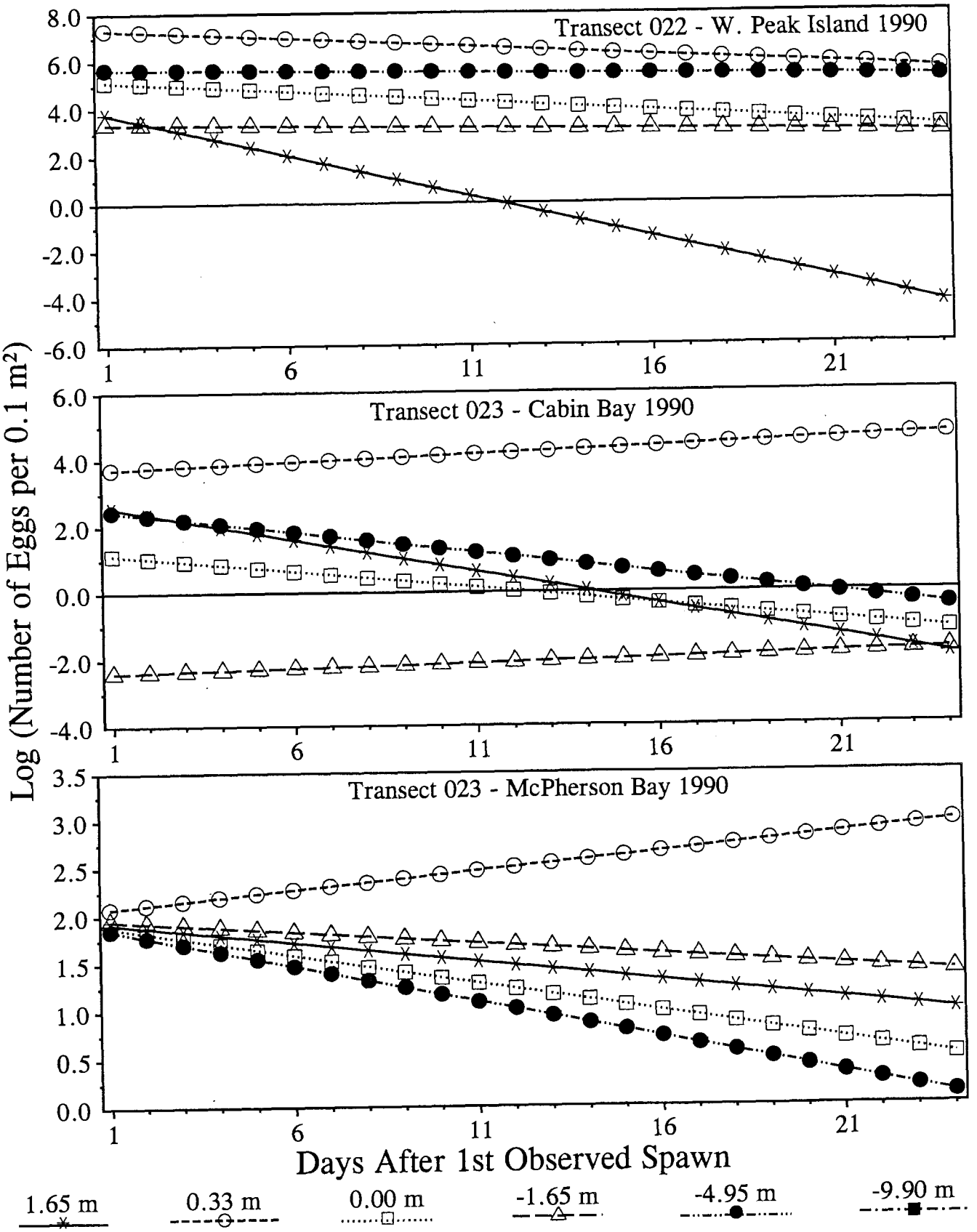


Figure 32. Estimated egg loss, by depth, at transect sampling locations in W. Peak Island, and Cabin and McPherson Bays, Naked Island area, Prince William Sound, Alaska, 1990.

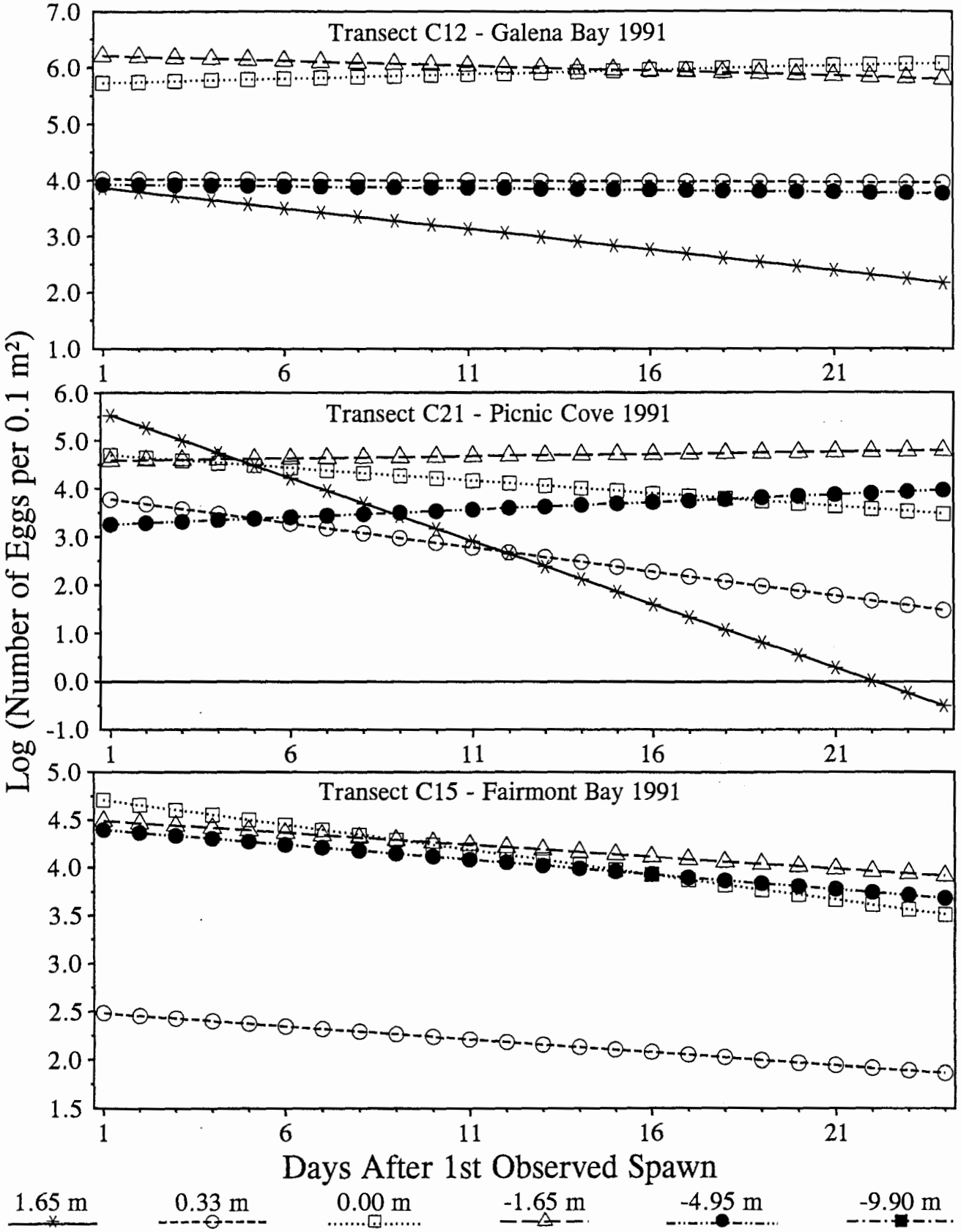


Figure 33. Estimated egg loss, by depth, at transect sampling locations in Galena Bay and Picnic Cove (Northeast area), and Fairmont Bay (North Shore area), Prince William Sound, Alaska, 1991.

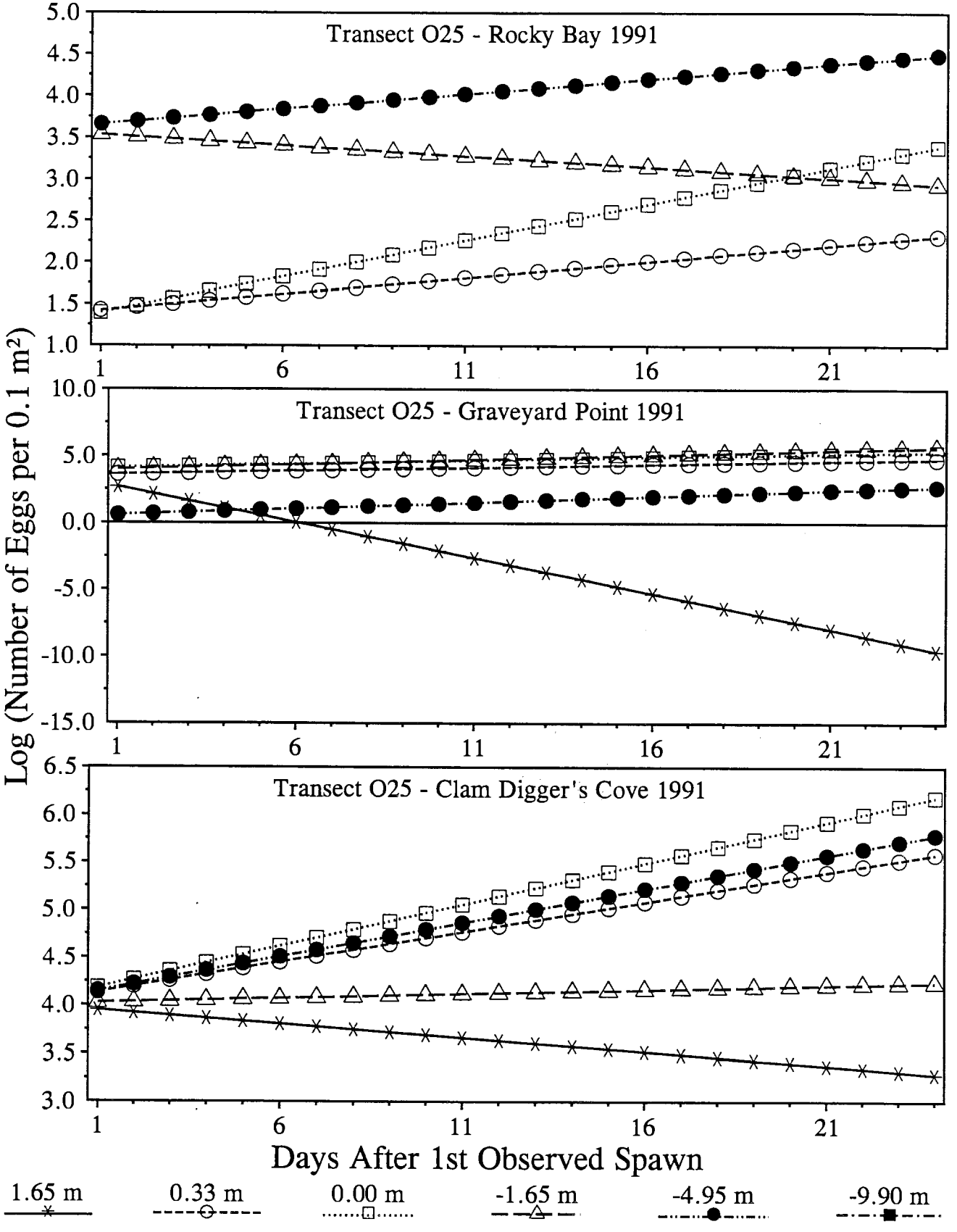


Figure 34. Estimated egg loss, by depth, at transect sampling locations in Rocky Bay, Graveyard Point, and Clam Digger's Cove, Montague Island area, Prince William Sound, Alaska, 1991.

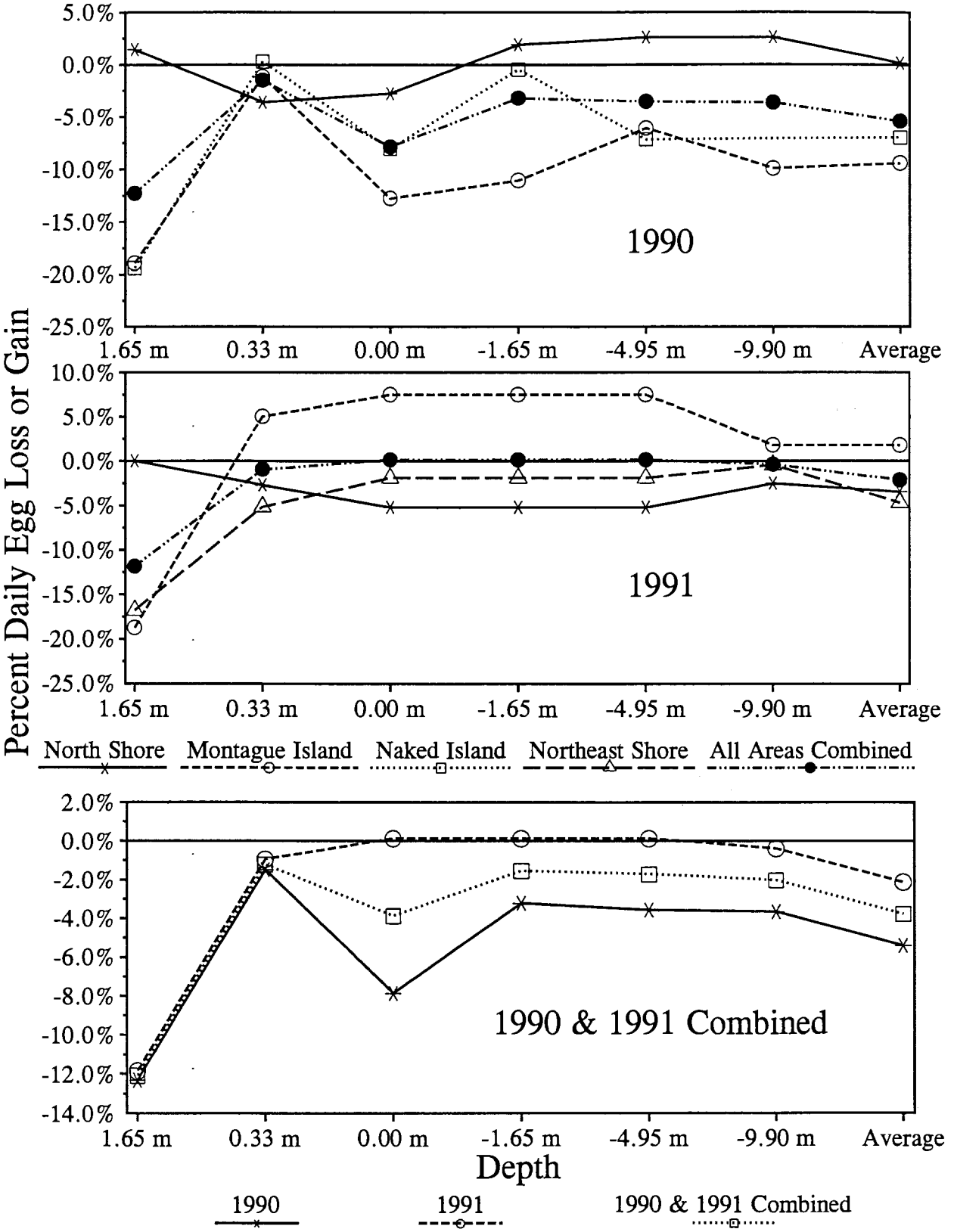


Figure 35. Estimated percent daily loss or gain, by depth, of Pacific herring eggs in Prince William Sound, Alaska, 1990-1991.

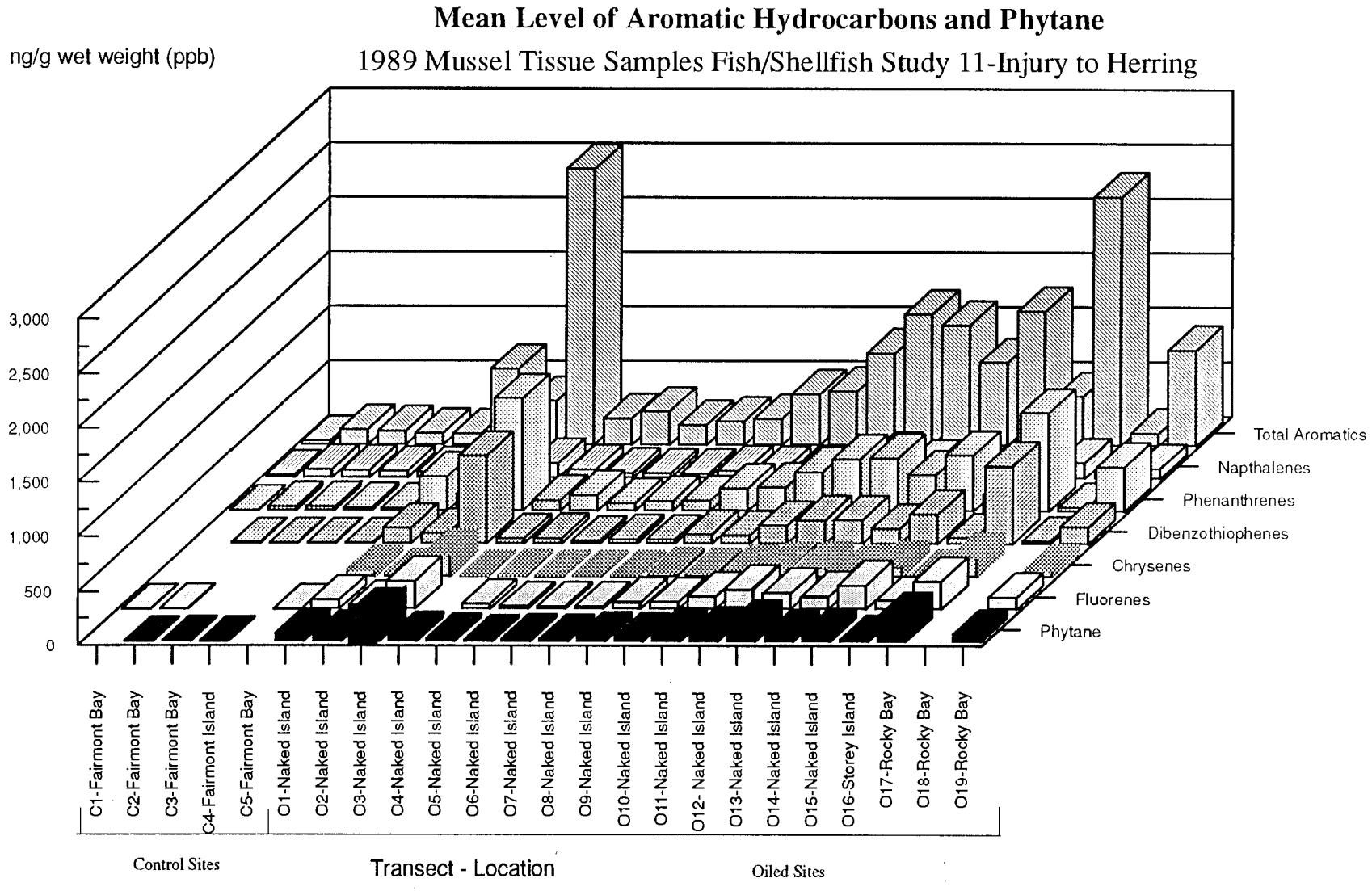


Figure 36. Mean level of aromatic hydrocarbons and phytane in mussel samples collected at transect sampling locations for Fish/Shellfish Study 11, 1989.

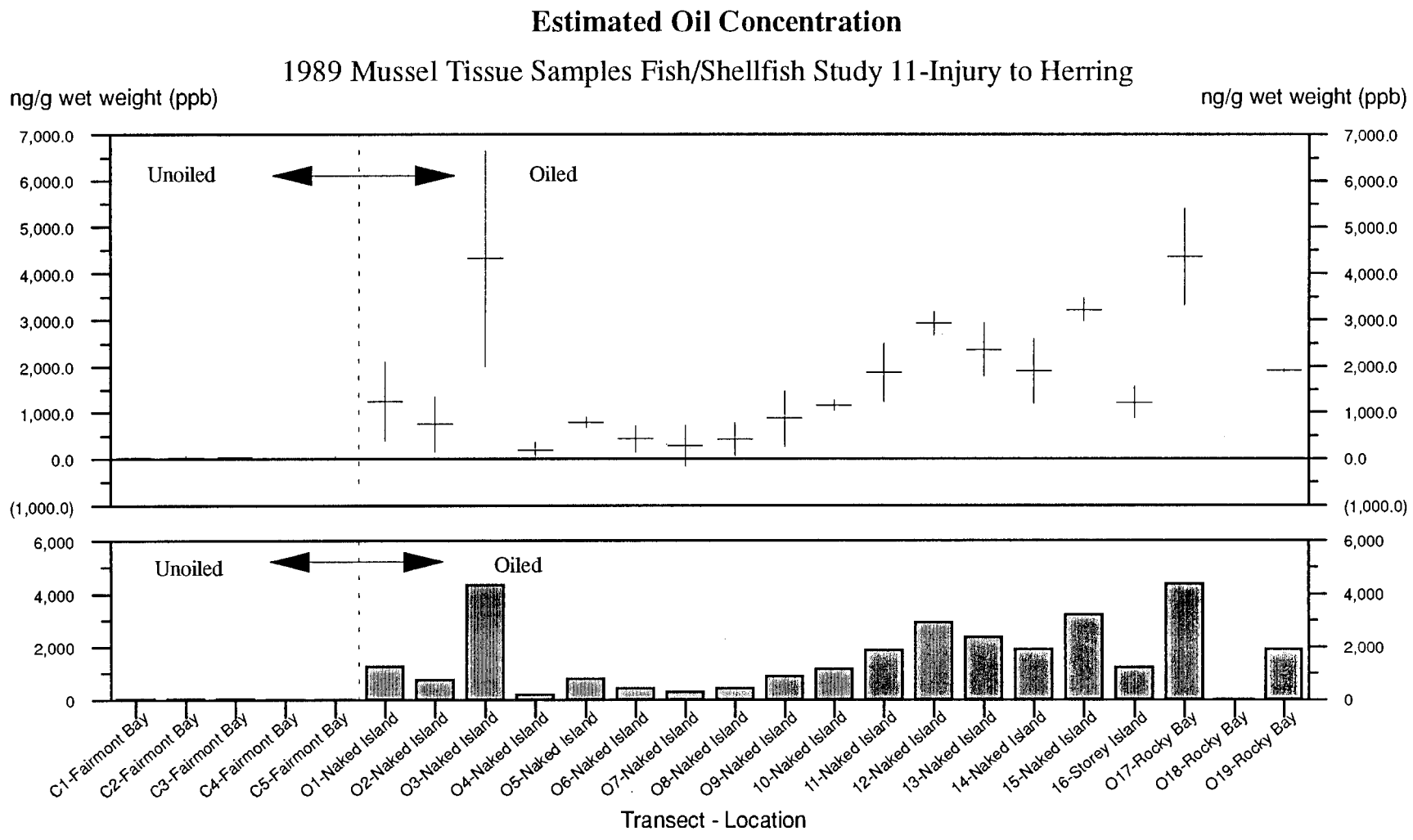


Figure 37. Estimated oil concentration in mussel tissue samples collected for Fish/Shellfish Study 11, 1989.

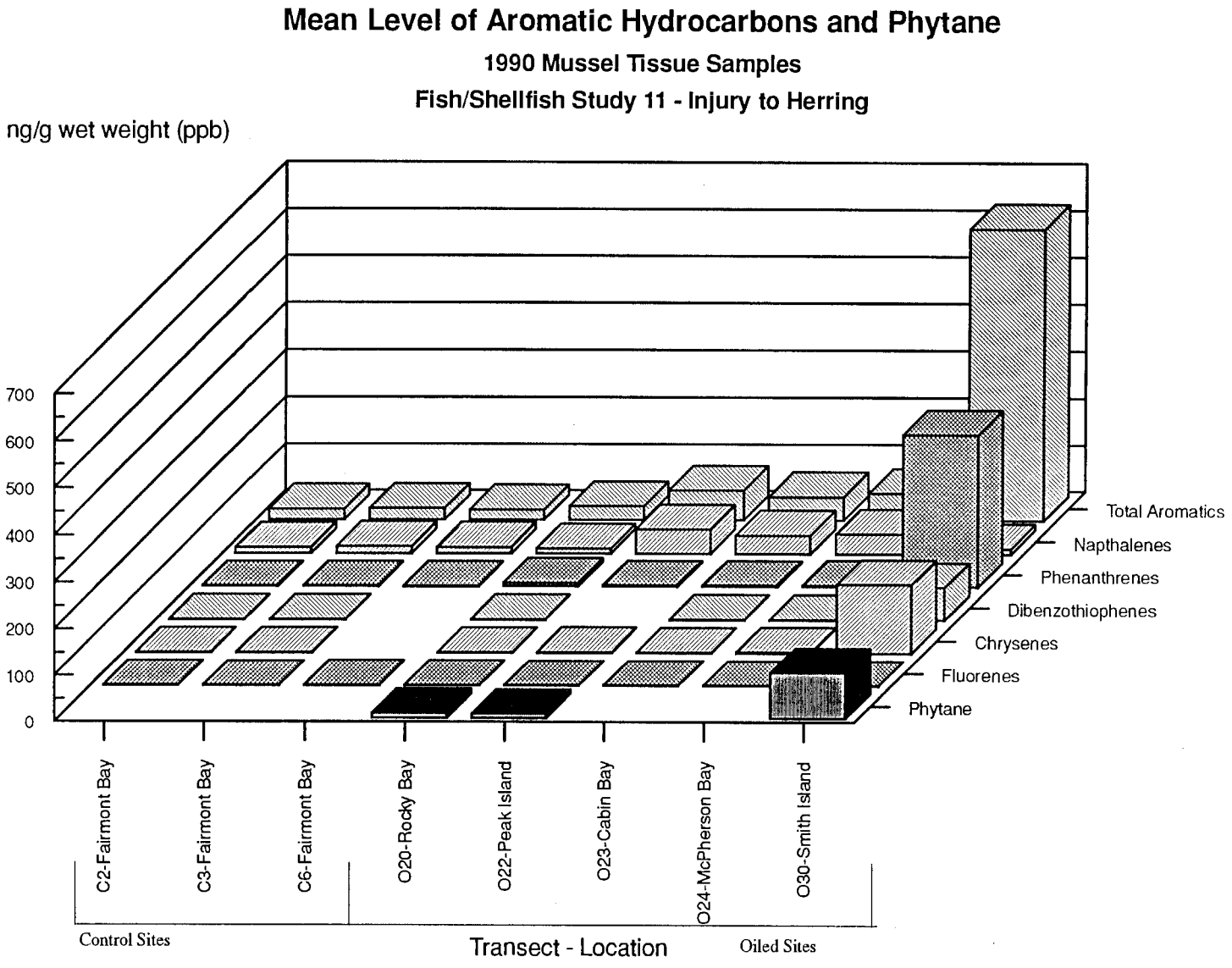


Figure 38. Mean Level of aromatic hydrocarbons and phytane in mussel tissue samples collected for Fish/Shellfish Study 11, 1990.

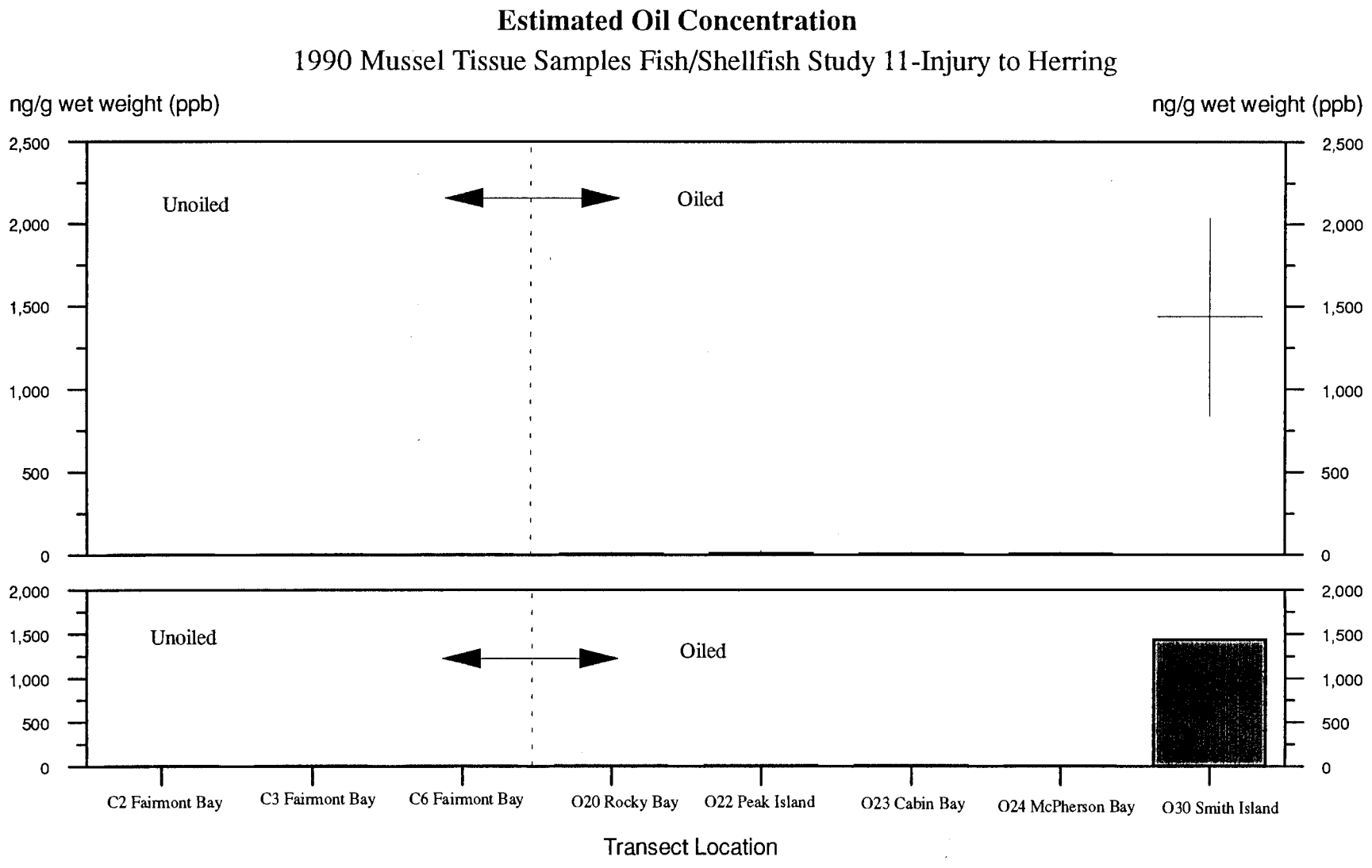


Figure 39. Estimated oil concentration in mussel tissue samples collected for Fish/shellfish Study 11, 1990.